The economic & environmental potential of biochar: A "win-win" solution for China's straw?

Abigail Jane Clare

Thesis submitted in fulfilment of
the requirements for the degree of
Doctor of Philosophy
to the
University of Edinburgh
May 2015

Declaration

I declare that this thesis has been composed solely by myself and that it has not been submitted, either in whole or in part, in any previous application for a degree. Except where otherwise acknowledged, the work presented is entirely my own.

Abigail Jane Clare

May 2015

Lay Summary

China is a rapidly developing country, and its agricultural sector is undergoing major changes. The majority of China's arable land is managed in very small plots by smallholder farmers, averaging under one hectare of land per household. As the cities develop, many working-age adults are migrating from rural to urban areas in search of higher incomes, leaving the older generation and their children behind on the rural family farms. This migration has two key impacts. Firstly, it reduces the availability of on-farm labour, and secondly it increases household income level (as the migrant workers send remittances home to the farm).

This has important consequences for the management of agricultural biomass, particularly straw resources. A lack of labour means that labour-intensive activities such as straw removal from the field and composting are increasingly difficult to complete. Moreover, higher incomes mean that straw is no longer needed as a source of fuel for heating or cooking. Thus, straw is increasingly becoming a waste product, and is regularly burned in the field causing significant local pollution problems.

This thesis therefore investigated the economic and environmental benefits of alternative uses for this straw, specifically either turning the straw into biochar or

into bioenergy.

Biochar is the name given to the charcoal-like substance produced when biomass, such as agricultural straw, is charred in the absence or limited presence of oxygen. When applied to soil, biochar can increase crop yields by making nutrients more accessible to plants, improving the ability of soil to hold water, and reducing the harmful acidity of soil that can occur when inorganic fertilisers are overused.

Another option for China's straw is to turn it into bioenergy. China has huge demands for energy and is increasingly trying to find sustainable, non-polluting forms to drive its economy. Therefore this thesis investigated the relative pros and cons of turning China's straw into biochar and bioenergy from economic and environmental angles. Specifically, it investigates whether there are "win-win" solutions from both of these perspectives.

This question was investigated from the perspective of smallholder farmers, commercial investors, and government priorities, using interviews conducted across four Chinese provinces, and secondary data collected from online and academic sources.

Overall the thesis finds that the economic and environmental benefits of using China's straw for bioenergy tend to outweigh the benefits of using China's straw to make biochar. This may be particularly true when using China's straw for co-firing applications in existing coal-fired power stations. However, the relative importance of bioenergy generation and soil fertility technologies such as biochar

may change over time, as populations grow and diets become more resource-intensive. Therefore biochar should not be entirely discounted as an option for China's straw resources in the future.

Abstract

Biochar has often been described as a “win-win” technology for soil fertility, agronomic yields, carbon sequestration and poverty reduction. However, despite a growing body of physical research evidence to support these claims, there is much less socio-economic evidence for biochar’s potential to achieve these “win-win” outcomes in real-world systems. Consequently, debates about biochar and its potential to contribute to sustainable development have often been polarised between extremes of opinion, with some claiming it is a key technology for mitigating climate change, and others warning of potentially dire effects for ecosystems and vulnerable populations. This inspired the objective for this PhD, which is to generate research that can inform and moderate the debate on biochar’s win-win potential. Guided by the theory of ecological modernisation, this PhD aimed to generate a body of applied, policy-relevant research on the economic and environmental potential of biochar as a win-win use of biomass resources. It was important to adopt geographical and biomass boundaries for the research to provide a meaningful and focused contribution, therefore the research is focused on China and its agricultural straw residues.

One of the central claims for biochar is that it can improve crop yields and, consequently, reduce poverty for smallholder farmers. This thesis investigated this from a socio-economic perspective using farm-scale linear programming models with primary data from interviews conducted across four contrasting Chinese

agricultural systems. The results suggest that biochar is unlikely to provide even minor economic gains, let alone poverty-reducing change, to smallholder farmers in these systems.

If biochar is not economic for farmers, there is a possibility that economies of scale made possible by business ventures could reduce the marginal costs per unit of biochar product and/or that governments/climate finance institutions may be interested in subsidising this technology where it has significant carbon mitigation impacts. Thus the next research question was whether biochar might be a profitable investment for businesses in China, and further whether businesses might also profit from carbon credits/subsidies where biochar's carbon sequestration potential is valued either by carbon markets or by climate conscious governments willing to provide appropriate incentives. Life-cycle and cost-benefit-analyses demonstrated that, when compared to the main competing uses for straw feedstocks (briquetting for combustion in boilers, and gasification for electricity generation), pyrolysis of straw to produce biochar makes a financial loss under all subsidy scenarios considered, and is the least cost-effective technology for carbon sequestration. Overall it seems biochar made from China's straw feedstocks is not currently a win-win option for smallholder farmers, business investors or national/international climate mitigation strategies.

In light of the relative dominance of bioenergy over biochar production as a financial and climate mitigating option for China's straw, the focus of the thesis shifts to explore win-win scenarios in this domain. Here the results are more promising. Combining a unique geographical dataset of China's coal fired powerstations and straw location with data on energy economics, the model suggests a small tweak to China's bioenergy subsidy system (an extension of the existing feed-in-tariff to include low energy replacement ratio cofiring) could contribute 42-62% of China's 2020 target to install 30GW of renewable energy

generation capacity: a classic win-win scenario for the Chinese government's bioenergy targets, bioenergy investors and global climate change.

Overall this thesis offers two main findings to the literature. Firstly it demonstrates that, within its current high application rate model, biochar will struggle to compete as a win-win strategy when viewed through financial and carbon sequestration lenses. However, secondly, it suggests that win-win strategies are available for China's straw resources under cofiring bioenergy applications. The thesis concludes with a critical discussion of these results in relation to the theory of ecological modernisation and the concept of win-wins.

Acknowledgements

Where to begin! Without question, these have been some of the best years of my life. I have had an unrivalled opportunity to explore fascinating subjects, cultures and countries, for which I am exceedingly grateful.

The journey has been challenging and rewarding in equal measure, but it wouldn't have been half so wonderful without some very special people. Firstly, thank you to my supervisors. To Simon Shackley for being an endless source of interesting ideas, perspectives, and grounded humility in the face of a very complicated world, and to Andrew Barnes for his solid advice, wry humour and "just do it" attitude, which got me through many a quantitative conundrum.

To my fieldwork friends. To Meng Qing-Nan, for five months of patient translation and friendship in the face of my awful Chinese ("would you like to do a donkey this evening?"), pestering questions, and dogged persistence that we do "just one more" interview. To Brad and Bonnie, for travelling 18 hours on the most awful bus to stay in a remote village and do questionnaires in the face of sparse accommodation, rotten chicken heads for dinner and, (apparently) the chance of being kidnapped by marauding gangs. My enduring gratitude for your *joie de vive* and positive attitudes. To Grace Wanjiru Kibue, to whom I owe more than it's possible to communicate. In the darkest times of total incapacity in China, you cooked, you shopped and you kept me in the greatest company I could have asked for. Without your unfailing help and companionship, I would have come home without data, and for that this "buggah" thanks you from the very bottom of my heart.

To Stephen Joseph, Genxing Pan, Chun-Ying Ji, Afeng Zhang, Lin Yang, and staff at the Kunming Institute of Botanics and the Sanli New Energy Factory: thank you so much for helping me to achieve my research aims, for the incredible dinners, and for the many, many bottles of *bai jiu*.

To John McDonagh, Dominic Moran, Jia Li, Andy Cross, Saran Sohi, Jim Hammond, Ya-Qing Gou, Luke Smallman, Michael MacLeod, Ron Wilson, Anthony Bloom and colleauges at the UK Biochar Research Centre in Edinburgh: thank you for your generosity of advice, willingness to teach, and for opening doors to ever more interesting projects. This PhD experience has been so much richer and varied for your inputs, and for that I am also very gratetful.

To Derek & Maureen Moss for their generous and heartfelt support for my various projects and ideas, from Northumberland to Ghana and back again! It has been a real pleasure to spend so much time in your company, and I thank you very sincerely for extending your knowledge, kindness and resources to me and to your many other students.

To Bron, Emily, Emma, Iain, James, Jess, John, Kathleen, Luke, Rach, Richard, Robyn, Sam, Tom, Qing and the rest of the Crew coffee room gang: it's been an absolute blast. It has been a delight to share this process with such an interesting, witty and empathic group of people. Long may our friendships continue!

To Kevin Randall, my step-father, for a constant supply of hearty dinners, generous support, and entertaining stories.

And finally, thank you to my parents. To Roy Clare, for inspiring the spirit of adventure in me, for teaching me stoicism in the face of discomfort, and for encouraging me to question the status quo. I know you would be proud to "have a doctor in the family". And to Jane Randall, for being my best friend, my greatest supporter, and intrepid co-adventurer. You mean everything to me.

Contents

List of Tables

List of Figures

Chapter 1

Introduction

1.1 A history of biochar

The term "biochar" is a relatively recent creation. It first appeared in a 1999 scientific paper (Bapat *et al.*, 1999) and was used to distinguish an activated carbon substance made from plant material feedstocks (in this case, sorghum) as opposed to activated carbon made from coal. However, the concept of creating stable carbonaceous material from crop residues as a large-scale climate mitigation strategy first appeared earlier in the 1990s (Seifritz, 1993; Somebroek, 1993) whilst research on this material as a soil amendment dates back to 1960s research on Amazonian dark earths (known as *"terra preta"*), which are estimated to be hundreds to thousands of years old (Neves *et al.*, 2003; Sombroek, 1966). These anthropogenic soils are thought to be created by the repeated addition of ashes and char (alongside other organic wastes) from the low-heat smouldering fires used by domestic populations, rather than by natural fires or slash and burn activities (Smith, 1980; Woods and McCann, 1999). These repeated organic additions create soils with up to 70 times more black carbon than surrounding soils, with an estimated mean residence time of 2,000 years (Kuzyakov *et al.*, 2009). This not only increases soil fertility, but also creates significant potential for carbon (C) sequestration, with total C storage reaching a maximum of 250Mg C ha^{-1} m^{-1} compared to 100Mg C ha^{-1} m^{-1} for typical Amazonian soils of the same parent material (Glaser *et al.*, 2001).

Combining these fields of research, it is estimated that the term "biochar" was first used in relation to both climate change mitigation and soil amendment at a March 2005 presentation by Johannes Lehmann (Woolf *et al.*, 2010a), entitled "Bio-char sequestration in soil: A new frontier" (Lehmann *et al.*, 2005). This was followed by a publication in 2006, defining biochar as a carbon-rich product that is created when biomass is heated in a closed container with little or no air available, and distinguishing it from charcoal by its intended use as a soil conditioner rather than as a fuel (Lehmann *et al.*, 2006).

1.2 The growing interest in biochar

This interest in biochar coincided with an increased awareness of agriculture as a significant contributor to global greenhouse gas (GHG) emissions (Robertson, 2000). Annual GHG emissions from agricultural production in 2000-2010 were estimated at 5.0-5.8 $GtCO_2e$ (gigatonnes carbon dioxide equivalent) per year (Smith *et al.*, 2014), with emissions from agriculture contributing around 13% of total anthropogenic emissions. Importantly, the AFOLU (agriculture, forestry and other land use) sector offers mitigation opportunities both through avoided emissions but also through enhanced removals of GHGs from the atmosphere, and biochar's carbon sequestration potential makes it an attractive climate-friendly agricultural technology. For example, research by Woolf *et al.* (2010b) suggests that the global technical carbon abatement potential of biochar could reach 6.6GtCO_2e per year, which exceeds the total estimated annual emissions from agricultural production during 2000-2010 presented above (Smith *et al.*, 2014).

Moreover, pressures on agricultural land are ever increasing, as this finite resource must meet the food, feed and fuel requirements of a growing population with increasingly resource-intensive consumption patterns (Godfray *et al.*, 2010; Smith *et al.*, 2010; The UK Government Office for Science, 2011). With almost all cultivatable land either occupied by tropical forests or already in use (Bruinsma *et al.*, 2003), emphasis has shifted towards the intensification of crop production, both through increasing the yields per unit area of land and, to a lesser extent, increasing the number of crops grown per seasonal cycle (Smith *et al.*, 2010). However, with global energy demand still predominantly met by fossil fuels, and with modern agriculture being highly dependent on these energy sources for its management and inputs (Woods *et al.*, 2010), there is an additional pressure for the intensification process to be environmentally sustainable (Tilman *et al.*, 2011; Garnett *et al.*, 2013). These pressing global issues have combined to form a "food, energy and environment trilemma" (Tilman *et al.*, 2009), in which the world's energy, environment and food challenges are heavily interconnected and require rapid and substantial change in order to meet global demand in the coming decades, whilst simultaneously avoiding severe environmental degradation (The UK Government Office for Science, 2011).

In the midst of these global challenges, the unique, multi-faceted properties of biochar make it a very interesting technological prospect for policy makers and scientists (Haefele, 2007; Lehmann, 2007; Lehmann and Steiner, 2009; Atkinson *et al.*, 2010). From a climate mitigation perspective, a combination of historical and experimental evidence suggests that production and application of biochar to soil could lock up sizeable stores of carbon for hundreds to thousands of years (Glaser *et al.*, 2001; Neves *et al.*, 2003), and from an agricultural perspective, biochar field trials have demonstrated a range of soil fertility and yield improvement impacts (Kimetu *et al.*, 2008; van Zwieten *et al.*, 2010; Jeffery *et al.*, 2011). Moreover the greatest yield impacts are most likely after biochar application to highly weathered soils with low soil organic matter content, cation-exchange capacity and pH (Kimetu *et al.*, 2008; Major *et al.*, 2010; Jeffery *et al.*, 2011; Spokas *et al.*, 2012). As these poorest agricultural soils tend to coincide with high rural poverty in developing countries (Sanchez, 2002; Crane-Droesch *et al.*, 2013), there seemed also to be opportunities for biochar to contribute towards poverty alleviation. This was further supported by the fact that biochar can be made from a wide range of agricultural residues, either purposefully in low-tech kilns or as a by-product of cooking using pyrolytic Improved Cook Stoves (ICS) (Whitman *et al.*, 2011; Shackley *et al.*, 2011a).

1.3 Biochar: the wins and the threats

Overall, biochar seemed to offer a variety of "wins" in agronomy, climate change mitigation and poverty alleviation, and the idea of biochar as a multi-win technology proved very attractive. Research publications on biochar increased from an annual 10-15 between 2002-2005 up to over 250 in 2013 (Lehmann and Joseph, 2015), many of them emphasising biochar's win-win properties. For example, Lehmann (2009) authored a paper entitled *"Biological carbon sequestration must and can be a win-win approach"*; Laird (2008) also outlined a *"Charcoal vision"* as *"a win-win-win scenario for simultaneously producing bioenergy, permanently* *sequestering carbon, while improving soil and water quality"*, and BBC writers even called biochar a *"win-win-win-win-win situation"* (Black, 2010).

However, despite this optimism, early studies also revealed that biochar is an extremely heterogeneous material, whose intrinsic properties and subsequent soil and crop impacts are affected by the type of feedstock material, production technology/conditions, the crop type, co-additions, and soil-type to which it is applied (Lehmann and Steiner, 2009; Jeffery *et al.*, 2011). This makes it particularly difficult to make generalised assertions about biochar's agricultural impacts, with field trials reporting both significantly positive (Glaser *et al.*, 2002; Major *et al.*, 2010; Cornelissen *et al.*, 2013) and negative (Chan *et al.*, 2008; Asai *et al.*, 2009; Jeffery *et al.*, 2011) impacts on crop yields. Moreover, the extrapolation about biochar's functions based on *terra preta* studies has also been questioned, as critics argue that *terra preta* soils were created over long periods of time within small-scale, biodiverse farming systems, and in combination with many other forms of organic materials (Ernsting and Smolker, 2009). This, arguably, bears limited resemblance to the high-application rate biochar field trials (typically ranging from 5-40t ha^{-1}) that have typified modern-day biochar agronomic trials.

Criticism of biochar has also spread beyond the uncertainties in predicting its agronomic impact, to the broader threats that biochar production could pose to land, ecosystems and people, fuelled by publicised plans to produce biochar on a commercial-scale from trees (Jha, 2009; Goodall, 2010), to create dedicated biomass plantations for biochar feedstocks on marginal lands (Read, 2009) and to include biochar as a carbon-offset mechanism within global carbon trading schemes. This, it was argued, would reduce the biodiversity of existing forest landscapes, increase competition for productive arable land, trigger land grabs

(Leach *et al.*, 2012), and displace pastoralists, hunters and gatherers from the so-called "marginal lands" where biomass plantations were intended to be created (Monbiot, 2009).

Overall these academic and popular debates around biochar highlighted two key research areas. Firstly, from the perspective of those perceiving biochar as a threat, there was the question of whether biochar could indeed incentivise the creation of large-scale biomass plantations, with negative consequences for biodiversity, food prices, and indigenous peoples interacting with marginal or degraded lands. Whilst this was a potentially fascinating research topic, these negative consequences were likely to occur at the scale suggested only if there were significant financial benefits to investors in backing biochar production, sale and application. However, cost-benefit analyses available at the time suggested that biochar would require carbon market support in order to be a viable economic option for farmers and businesses (Brown *et al.*, 2010; Roberts *et al.*, 2010; Shackley *et al.*, 2011b), and biochar seemed a long way from achieving carbon market accreditation, particularly after the dismissal of the Carbon Gold application for biochar to be accredited under the Voluntary Carbon Standard. As such, research on the validity of the proposed "threats" seemed less relevant than deeper investigation of biochar's proposed "win-win potential", and this is therefore the research area that this thesis focuses on.

Before outlining the specific "wins" that were investigated, the following section provides a theoretical and historical overview of the concept of "win-win" technology solutions for environmental management.

1.4 The theory of Ecological Modernisation

The concept of win-win technologies is inextricably linked to the creation of sustainable development as an approach to environmental management, which itself has significant roots in the Brundtland Report (World Commission on Environment and Development, 1987), which states that,

In essence, sustainable development is a process of change in which the exploitation of resources, the direction of investments, the orientation of technological development, and institutional change are all in harmony and enhance both current and future potential to meet human needs and aspirations (p.46).

The key message from this statement is that economic, technological and human progress can continue in ways that do not harm the earth and its ability to support future generations. Intrinsically this means that win-win solutions must be found that can advance this vision of development whilst having a neutral or positive impact on the environment. Today, this vision of sustainable development is the dominant high-level, international discourse around environmental management (Dryzek, 2013). However, the Brundtland Report marked a significant departure from other popular environmental discourses at the time, which tended to emphasise the limits to growth, looming resource scarcities and/or the need for a radical change in societal power structures and resource governance (Meadows *et al.*, 1972; Barney, 1980). In contrast, the sustainable development approach outlined in the Brundtland Report emphasises the need to find common ground and mutually reinforcing pathways between economic growth, environmental protection, population stabilisation, global and intergenerational equity. Therefore, for sustainable development to be achieved in practice, it follows that there must be "win-win" development pathways that can resolve the apparent conflicts between the world's economic growth trajectory and global environmental limits, whilst also ensuring social justice and intergenerational equity.

However, with such ambitious and broad aims, it has been particularly difficult for stakeholders to agree on a single definition of sustainable development. The most well-known definition from the Brundtland report (*"...development that meets the needs of the present without compromising the ability of future generations to meet their own needs"*) was not accepted by everyone. Questions were raised about what and whose needs count, who defines those needs, and on what time scale (Dobson, 1998), whilst some challenged the basic premise that growth can continue indefinitely, sustainable or not (Daly, 1990; Jackson, 2011). Others felt that defining sustainable development was not the central issue, and rather that the real problem lay in *"determining what has to be done to achieve it"* (Pearce, 1993).

In the context of these definitional difficulties, another related theory, known as ecological modernisation, was also gaining popularity during the 1980s. Ecological modernisation, like sustainable development, suggests that economic and environmental progress can continue together, however ecological modernisation has a narrower focus than sustainable development (i.e., not considering issues of justice and/or intergenerational equity), arguably making it easier to operationalise (Baker, 2007).

Ecological modernisation was initially developed by German scientists Huber and Jänicke as an interpretation of the shifts in environmental policy-making occuring in Germany, and it was used to explore societal attempts to respond to the negative environmental consequences of modernity, and also as a strategy through which countries could address the ecological problems created by industrialisation (Blühdorn, 2001; Baker, 2007). At that time in the 1980s, criticism of the "end-of-pipe" approach to environmental management was growing, recognising that it tended to displace problems in space and time, rather than adequately address them. Moreover, there was increasing awareness that environmental protection was not necessarily part of a zero-sum trade-off with economic prosperity, but rather that the integration of environmental concerns into indsutrial modernisation could be a positive-sum game, creating a source of future growth (Weale, 1992).

Although ecological modernisation is used in different ways by different authors (Christoff, 1996) there are four generally accepted themes (Baker, 2007). First, similar to sustainable development, ecological modernisation assumes that there are synergies between environmental protection and economic growth, and that states will have a strong role to play in improving industrial policy and stimulating research and development to find these synergies. Second, states must integrate the environment into a broad range of government policy-making areas, thus ensuring "environmental policy integration" (EPI). Thirdly, new environmental policy instruments will be needed to achieve this integration and, finally, these state-driven changes will occur through sector-specific activity, with an emphasis on the invention, innovation and diffusion of new technologies within the industrial sector.

Thus, ecological modernisation shares some similarities with sustainable development in its anthropocentric approach to environmental management, and their search for synergies between economic and environmental progress (Jänicke, 2008;

Dryzek, 2013). However, although ecological modernisation and sustainable development both search for "win-win" synergistic solutions, they also differ, most notably in the scope of issues that they address within these win-wins. For example, whilst the Bruntland Report places issues of social justice and intergenerational equity at the centre of its sustainable development definition, ecological modernisation remains silent on these issues (Langhelle, 2000). Moreover, ecological modernisation arguably has a more local frame of reference to its issues. For example problems of water pollution, chemical waste and acidification fit easily within its framework (Mol, 1996), whereas broader issues of global warming and biodiversity conservation are much harder to address using the ecological modernisation approach (Mol and Spaargaren, 1993). Some authors therefore see ecological modernisation as a narrower sub-set of sustainable development, or even as a necessary but not sufficient strategy for achieving sustainable development (Dryzek, 2013; Langhelle, 2000). However others warn against the "seductive appeal" of ecological modernisation, which suggests that environmental problems can be remedied without the need for substantial changes in our patterns of consumption or distribution of wealth (Baker, 2007), whilst others worry that adoption of ecological modernisation as the dominant approach to environmental protection will stifle the (they believe, necessary) transformative potential of environmental movements and may delay or exacerbate major environmental crises (Giorgi and Redclift, 2000).

1.5 Biochar and Ecological Modernisation

In many ways, the early descriptions of biochar as a win-win technology typify the industrial innovations that are sought within the theory of ecological modernisation. Firstly, biochar was described as a systems-level, preventative technology, rather than a curative, end-of-pipe approach. (Although there have been suggestions for biochar as an end-of-pipe geoengineering solution (Read, 2008; Glaser *et al.*, 2009; Lenton and Vaughan, 2009) there have also been acknowledgements of the risks inherent in this course of action (Downie *et al.*, 2012; Leach *et al.*, 2012). Overall the dominant research and popular narrative is of biochar being adopted as a part of a transition towards climate-friendly agricultural practices (Cernansky, 2015; Lehmann and Joseph, 2015).) Biochar was also framed in terms of its win-win potential to contribute both to economic development (by increasing agricultural yields) and also to environmental management (by sequestering carbon and/or reducing soil nitrate leaching), without necessitating a radical change in the structure or operation of the existing capitalist system. Indeed, biochar has often been couched in terms that are central to capitalism, with various studies investigating its profitability for farmers and businesses (Brown *et al.*, 2010; Roberts *et al.*, 2010; Galinato *et al.*, 2011; Shackley *et al.*, 2011a; Field *et al.*, 2013; Kung *et al.*, 2013) and discussing its suitability as a technology through which carbon credits might be traded (Lehmann, 2007; Whitman and Lehmann, 2009; Pratt and Moran, 2010). Finally, it has become increasingly clear that biochar will require significant state/institutional support through direct valuation of its environmental impacts, further research & development, and agricultural/environmental policy making in order to be widely adopted by farmers (Shackley *et al.*, 2015). This clearly echoes the ecological modernisation assertion that state involvement will be necessary to re-direct economic growth on a sustainable pathway.

Moreover, the early criticisms levelled at biochar are also similar to those directed at ecological modernisation. For example, biochar was seen as a distraction from the broader issues of over-consumption (Ernsting and Smolker, 2009), similar to the warnings about ecological modernisation's *"sedutive appeal"* (Baker, 2007).

Moreover, biochar was criticised for its potential negative impacts on vulnerable populations, through land grabs, increasing food prices or displacement of native populations (Monbiot, 2009; Leach *et al.*, 2012). These critiques are focused on the social justice and equity implications of biochar deployment, which are areas that the theory of ecological modernisation notably does not address (Langhelle, 2000).

Overall, biochar fits well within the goals and scope of ecological modernisation and this theory is therefore used as the guiding framework of reference within which to assess biochar's win-win potential. The following section therefore discusses the evidence that was available to support biochar's win-win potential in late 2011, when this research was being planned, and identifies the research gaps that are investigated in this thesis.

1.6 Biochar: the research gaps

In late 2011, when this PhD began, there were many claims being made about biochar's multi win potential, with the central two "wins" being biochar's agronomic benefits and its ability to store carbon in soil, thus contributing to climate change mitigation. These wins also map well onto those that are sought in ecological modernisation, namely that a technology will contribute towards economic progress (in the case of biochar, through improving agronomic yields and/or fertiliser use efficiency) whilst also improving environmental management (in this case, mitigating climate change). However, the research evidence available to support these win-win claims came predominantly from the physical sciences, with very little supporting evidence from the social sciences.

Figure 1.1 outlines the suggested economic and environmental wins, the evidence

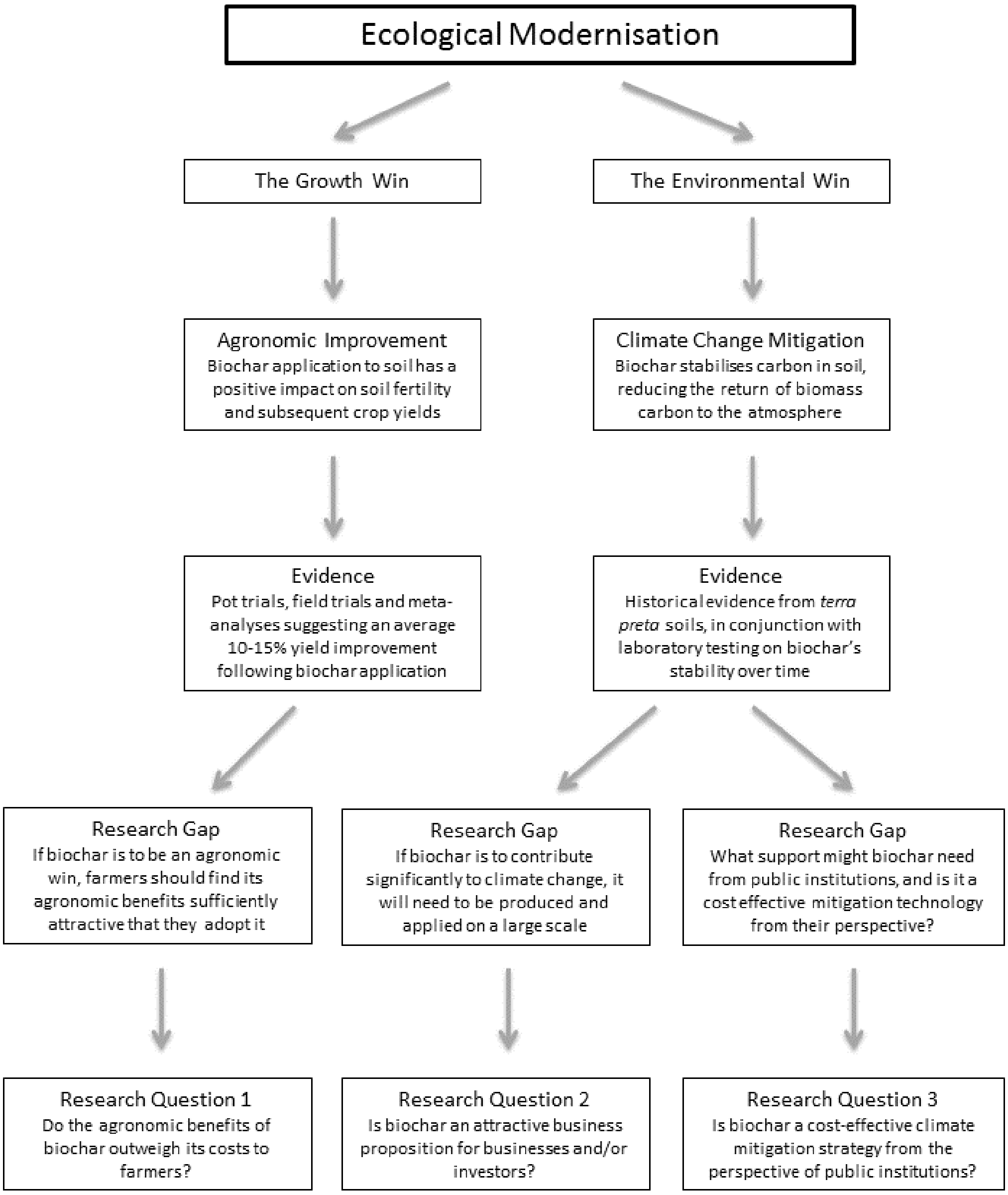


Figure 1.1: Biochar's win-wins, evidence, research gaps and questions

that was available to support these claims in late 2011, and the subsequent research gaps and three research questions that were initially identified for this thesis.

Research question 1 investigates whether the agronomic wins that field trials and pot trials were demonstrating would be sufficient to outweigh the costs and benefits of biochar sourcing and application. Although cost-benefit analyses were emerging in late 2011 (Brown *et al.*, 2010; Roberts *et al.*, 2010; Galinato *et al.*, 2011), these had not considered the perspective of small-scale farmers, making biochar on their farms using available agricultural residues. Moreover, there was no academic research on biochar's potential as a poverty alleviation strategy (which had also been outlined as a potential "win"), therefore this research question also included this as part of its scope.

Research question 2 considers the scale at which biochar might be adopted, which is important when considering that biochar will need to be produced on a large-scale in order to contribute to climate change mitigation. For this technological scaling to happen, business investment will be necessary, and biochar will need to be both profitable as an investment opportunity in its own right, and also profitable in comparison to alternative commercial uses of biomass feedstocks. This research question therefore investigates the costs and benefits to businesses of producing and selling biochar compared to alternative uses of biomass feedstocks.

Finally, research question 3 investigates whether biochar is a cost-effective climate change mitigation strategy. Specifically, if biochar was to require financial support to achieve large-scale adoption and subsequent climate mitigation services, how much support would it require per tonne of CO_2e sequestered, and how does this compare to alternative uses of the biomass feedstocks that would be used for biochar production?

1.7 Choosing a geographical boundary

Having placed biochar within the frame of ecological modernisation and outlined the specific “wins” to be investigated, it was important to develop a geographical boundary for this research. The choice of a developing country was natural, as research on biochar’s socio-economic win-win potential up to 2011 had only been conducted in developed country case-studies, and the assertion that biochar may assist in poverty alleviation had not yet been studied academically. Moreover, research was emerging that biochar has the greatest agronomic impact on acidic, degraded soils (Kimetu *et al.*, 2008; Jeffery *et al.*, 2011), which tend to be located in developing countries (Sanchez, 2002). Thus a study of biochar’s “win-win” potential within a developing country context was logical and justified.

From the perspective of ecological modernisation, China presented itself as an ideal case study country. In 2007, the Chinese Academy of Sciences (a respected Chinese research institution with strong links to the government and major media impact) released a 450-page high-profile report on ecological modernisation theory in China (China Centre for Modernisation Research, 2007), which was followed by high level statements from Premier Wen Jiabao, calling for more sustainable economic growth for the country, and a “leaner and greener” China (Zhang *et al.*, 2007). This marked the first official inclusion of the environmental domain into the Chinese perception of modernisation, which had previously been dominated by concerns such as modernising agriculture, industry, national defence, science and technology in the 1960s; solving the problem of inadequate food and clothing in the 1970s; and the coordinated modernisation of economy, society, politics and culture in the 1990s (Zhang *et al.*, 2007).

This inclusion of the environment in China's modernisation priorities signifies the greater importance that the Chinese government is placing on protecting its natural resources. For example, in light of rapidly worsening environmental conditions across the country, China's State Environmental Protection Administration (SEPA) launched a programme of "environmental storms" between 2005-2007, trying to address the environmental damage caused by China's narrow focus on economic development during the previous two decades (Zhang *et al.*, 2007). In addition, China launched the world's most ambitious "green GDP" measuring project from 2004-2006, attempting to develop a measure of GDP that included a discount related to the negative environmental impacts that arise from economic activity (Li and Lang, 2010). Alongside these projects, the *2007 China Modernisation Report* can be seen as an attempt to integrate environmental concerns and ecological protection into economic policies, moving it from an "end-of-pipe" to a systems-level approach.

In addition to supporting the ecological modernisation framework, China offers a vast range of agro-ecosystems within which to investigate, contrast and compare the economic and environmental impacts of biochar. China houses over 200 million smallholder farmers (Huang *et al.*, 2012b), living in varying stages of socio-economic development across a large geographical area, covering a huge variety of climates, crops, and agro-ecosystems. This offered the possibility to investigate the socio-economic suitability of biochar across a wide range of farmer types, thus ensuring that interests of smallholder subsistence farmers all the way up to large-scale commercial farmers could be considered.

Finally, with multiple active biochar research groups, numerous pyrolysis units built/under-construction and some nascent biochar businesses being set up, China was likely to be a data-rich environment within which to investigate biochar's socio-economic potential from the perspective of multiple stakeholders. The following section provides more details on China's agricultural history, and further expands upon why biochar could theoretically address a number of the environmental and biomass management challenges facing Chinese farming today.

1.8 Agriculture in China

Following the Rural Reforms instigated by Deng Xiao Ping in the years following the death of Mao Tse-tung, agricultural productivity in China increased dramatically. Starting in 1979, Deng steadily dismantled the communal farming system and introduced the Household Responsibility System (HRS), under which households were allotted parcels of agricultural land according to their family needs and labour availability. Following these reforms, grain production increased by an impressive 4.7% per year from 1978-1984, despite the sown area not changing in size (Huang and Rozelle, 2015). In addition to increasing grain production, China has also been moving towards higher-value agricultural goods, such as horticultural, livestock and aquaculture products: between 1990 and 2005 China's vegetable production capacity increased by the equivalent of California's production capacity every two years (Huang and Rozelle, 2015).

However, if the growth in agricultural productivity has been large, this has been dwarfed by productivity improvements of the industry and service sectors, which have grown at two to three times that of agriculture (see Table 1.1, modified from Huang *et al.* (2012b).)

As a result, the contribution of agriculture to China's gross domestic product (GDP) is shrinking, falling from 40% in 1970 to 12% in 2005 (Huang and Rozelle, 2015). This is reflected in the mass rural to urban migration that China has experienced in recent times, and also in structural changes to rural society and incomes. For example, although the output per unit of land and agricultural labour productivity have risen for rural households (Lin, 1992; Jin *et al.*, 2002), the share of net rural household income from agriculture has declined from 66.3% in 1985 to 29.1% in 2010 (Huang and Rozelle, 2015). Thus the contribution of agriculture to national GDP and household income has grown in absolute terms, but shrunk in relative terms to the industrial and service sectors.

Table 1.1: Annual GDP growth rates of China's economy

	1970-78	1979-84	1985-95	1996-00	2001-05	2006-10
GDP	4.9	8.8	9.7	8.2	9.9	11.1
Agriculture	2.7	7.1	4.0	3.4	4.3	4.5
Industry	6.8	8.2	12.8	9.6	11.4	11.9
Service	n.a.	11.6	9.7	8.3	10.1	11.9

Almost thirty years on from the start of the rural reforms, China has one of the world's most liberalised, reformist agricultural sectors, particularly compared to other developing countries (Rosen *et al.*, 2004). It has made deep cuts to agricultural tariffs and has committed to eliminating export subsidies. Moreover, despite having recently moved from taxing to subsidising grain and agricultural inputs, research suggests that these subsidies are non-distorting, because they are awarded on a per unit land basis and therefore do not appear to distort producer decisions (Huang and Rozelle, 2015).

China has also made significant steps to improve household property rights over agricultural land. Although agricultural land and the income from it is controlled by the households to which it is allocated, the land remains the property of the state and cannot be sold. Moreover, although land can be rented out during the tenure period that the state grants each household, an uncertain legal framework for renting has slowed the development of land rental markets (Deininger and Jin, 2005) by impeding the efficient transfer of land between households and the development of larger farm holdings. Recognising this, China has passed a series of laws (Land Management Law, 1998; Land Contracting Law, 2003; Rural Land Contract Law, 2003; Property Law, 2007) to increase the security of household tenure over land and clarify the rights for transfer and exchange of contracted land (Huang *et al.*, 2012b). To date there is evidence that both the number of households renting land in/out and the size of those rented land parcels are increasing (Huang *et al.*, 2012a) however it is likely that large-scale willingness of farmers to rent out land will depend on their alternative means of social security (Huang and Rozelle, 2015).

Overall, despite progressive reforms and major success in increasing agricultural productivity and rural incomes, Chinese agriculture faces serious challenges. Firstly, agricultural production is still dominated by over 200 million smallholder farmers, controlling an average land size of just over half a hectare per household, which itself is often divided into multiple smaller parcels of land. Around 60% of these plots are smaller than 0.1ha, with just a quarter over 0.15ha (Huang *et al.*, 2012b). On-farm labour is also decreasing, as working age adults migrate to urban centres in search of higher paid work, and this is leading to both a feminisation and an aging of the agricultural labour force (de Brauw *et al.*, 2012; Wang *et al.*, 2012b). This reduction in labour availability in combination with the reduced economic importance of farming to households is also contributing to wide-spread inefficient and environmentally damaging farming practices. For example, although Chinese cereal grain yields increased by 65% from 1980 to 2010, the use of chemical fertilisers during that period increased by 512% (Zhang *et al.*, 2011) and Chinese farmers now use more chemical fertiliser (an average of 200kg per ha) than farmers anywhere else in the world (Huang *et al.*, 2008). This excessive fertilisation (mostly from nitrogenous fertilisers) has resulted in serious environmental problems, including eutrophication of surface waters, nitrate pollution of groundwater, acid rain and soil acidification. Staggeringly, over half of China's lakes suffer from eutrophication (Jin *et al.*, 2005; Fan *et al.*, 2012).

Moreover, research on Chinese farming systems increasingly demonstrates that N application rates can be significantly reduced (up to 70% in some cases) without sacrificing crop yields (Huang *et al.*, 2008; Ju *et al.*, 2009) or even with a positive impact on crop yields (Zhang *et al.*, 2015). However, efforts to retrain farmers to use less fertiliser have so far proven both costly and ineffective, even where they could increase household income by up to 15% (Huang *et al.*, 2008; Guo *et al.*, 2015; Zhang *et al.*, 2015).

A further environmental challenge for China's agricultural sector is the widespread practice of on-farm straw burning. Each year China produces 800 million tonnes (Mg) of crop residues, of which an estimated 505 million Mg are available for use after retaining quantities of organic material to maintain soil quality (Jiang *et al.*, 2012). However a significant proportion of these residues, typically the straw portion, is burned in-field as a result of reduced demand for straw as a household fuel, a scarcity of on-farm labour/mechanisation for straw collection, and the imperative for increasingly time-poor farmers to quickly dispose of waste residues before planting the next crop (Wu *et al.*, 2001; Lin and Song, 2002; Yu, 2003; Cao *et al.*, 2008). Straw burning on this scale is an inefficient use of biomass resources and causes significant air pollution in both rural areas and nearby cities (Li *et al.*, 2008; Yang *et al.*, 2008), emitting high levels of particulate matter (PM), hydrocarbons and other pollutant gases to the atmosphere (Duan *et al.*, 2004; Yan *et al.*, 2006; Qu *et al.*, 2012). However, despite the Chinese government announcing a variety of straw burning bans since the late 1990s, enforcement has proven difficult, costly and ineffective (Jingjing *et al.*, 2001; Qu *et al.*, 2012). The high availability and apparent wastage of China's straw resources made this an ideal biomass source to research as a potential biochar feedstock within China, and this is therefore the focus biomass source for all analyses in this thesis.

Overall, many of China's agricultural challenges as outlined above could theoretically be addressed through biochar production and agronomic application. For example, the conversion of China's waste agricultural biomass into biochar could theoretically reduce air pollution from avoided open biomass burning, improve rural waste management, increase soil quality, increase crop productivity and reduce fertiliser leaching from the soil. Once again, these characteristics of the Chinese agricultural system made it an attractive setting within which to test biochar's socio-economic potential.

1.9 Thesis objectives

The goal of this thesis is to provide applied, policy-relevant research that investigates whether biochar can live up to the win-win claims made about it from the mid-2000s onwards. Specifically, Section 1.8 outlined three research questions, addressing the socio-economic suitability of biochar to farmers, the relative profitability of biochar to businesses/investors, and the cost-effectiveness of biochar as a climate change mitigation technology from the perspective of governments and/or global environmental actors. With the addition of a geographical research boundary (China) and specified biomass feedstock (agricultural straw residues), the thesis objectives are as follows:

Objective 1: To investigate whether biochar's agronomic benefits (defined as changes to net-farm profit from increased crop yields and/or reduced fertiliser use resulting from biochar application) can outweigh the costs of biochar sourcing and application (definfed as $ per kg biochar produced and applied) for Chinese farmers across a variety of farming systems

Objective 2: To determine whether biochar production and sale is a profitable investment opportunity (measured as net present value) for Chinese businesses, in comparison to alternative uses of agricultural residues

Objective 3: To explore the climate change mitigation impacts of producing biochar from China's crop straw residues compared with alternative uses of that biomass and, subsequently, to determine whether biochar is a cost-effective climate change mitigation strategy compared to these alternative biomass uses

These objectives were answered through research undertaken for Chapters 3 and 4, the results from which suggest that biochar will struggle to compete with bioenergy as a use of China's straw feedstocks, from both environmental and economic perspectives. In light of this information, an additional thesis objective was developed to explore the use of China's straw resources from a bioenergy perspective and to investigate how China's existing bioenergy policy landscape might be improved to ensure the most efficient use of its agricultural residues for bioenergy production. Specifically:

Objective 4: To explore how much bioenergy (terawatt hours (TWh) of electricity) can profitably be produced (at an internal rate of return of 8% or more) if China were to extend its feed-in-tariff for bioenergy to include low energy replacement ratio cofiring of agricultural residues in existing coal-fired power stations

1.10 An overview of the thesis chapters

This thesis is structured as a series of six chapters, three of which are based on published research papers (Chapters 3-5), each of which stands alone as an independent research article. These papers are presented in the chronological order in which they were produced, predominantly because their results build upon each other, reflecting the train of thought that developed throughout the PhD process, and that knits these three independent pieces into a coherent whole.

Following this introduction, Chapter 2 introduces the methodological approach that evolved from the applied-research objectives of the thesis and the adoption of ecological modernisation as a guiding theory.

The subsequent three chapters (Chapters 3-5) are the substantive research chapters. Chapter 6 brings the findings of the preceding chapters together and discusses the way they have advanced our understanding of win-win solutions for China's straw resources. It concludes with a critical discussion of the win-win concept and ideas for future research directions.

Chapter 2

Theory and Methodology

The aim of this thesis was to generate a body of work that provided a data-informed perspective on whether biochar could be a win-win technology from the perspective of ecological modernisation theory. Specifically two wins were identified for further investigation: the economic win (biochar's agronomic impacts) and the environmental win (biochar's climate change mitigation potential). The theory of ecological modernisation emphasises the idea that environmental management can occur in synergy with economic growth. However, as outlined in Chapter 1, there was very little published research available on the economics of biochar when this research was being planned. Therefore, economic analyses and methods were required to fill this research gap, and specifically cost-benefit analyses were needed in order to investigate biochar's relative attractiveness as an agronomic input for farmers (Objective 1), as an investment opportunity for businesses (Objective 2), and as a cost-effective climate change mitigation technology for governments and/or international environmental actors (Objective 3).

This thesis therefore draws upon traditional neo-classical economic theory as a framework within which to assess biochar's costs and benefits to different stakeholders. The following sections outline the history of neo-classical economic theory, and describe the linear programming, financial & social cost-benefit and life-cycle analysis methods that are used in this thesis.

2.1 Neoclassical economic theory

Neoclassical economic theory holds that the main factor affecting an individual's decision to undertake an action is the expected change in utility, or well-being, following that action. Each individual has a utility function that can be maximised when making certain choices. However, the utility of any given action will vary between different people, and as an intrinsic concept it is therefore difficult to measure accurately. As a result, studies of utility have historically used money as a proxy for the utility value that an individual expects from any given action, i.e., it is assumed that people will pay more money for something that gives them greater utility or, alternatively, that people will choose the greatest profit-maximising action when presented with a range of possible options. Applying this rationale to farmer behaviour, a farmer would be expected to adopt technologies that they expect will most increase their farm profits, and this theory has been used as the basis for many agricultural extension models that assume adoption of profit-maximising technologies to be inevitable where sufficient information is made available to farmers (Vanclay & Lawrence, 1994).

However, there are well-established criticisms of these assumptions. Firstly, it is increasingly clear through both economic theory and experimental evidence that humans do not behave as rational economic agents (Selten, 1990; Thaler and Sunstein, 2008; Kahneman, 2011). They can be strongly influenced by cognitive and behavioural biases, leading to choices that do not reflect the logic put forward in standard utility theory. Moreover, even where humans attempt to follow a logical path, they often have limited resources with which to make decisions, either in the availability of information or in their ability to process the vast complexity of it. For farmers this can be particularly pertinent, as factoring environmental factors, economics and risk into the decision to adopt a new technology requires significant experience, knowledge and cognitive capacity. Secondly, farmers do not necessarily respond to changes in farm profit, and the associated utility, in a uniform way. For example, some farmers adopt experimental technologies with limited proof of utility (so-called "innovators" or "first-users"), whereas others will only adopt a new technology once its utility value is proven (so-called "late adopters") or not at all ("non-adopters") (Rogers, 2010). Finally, it is clear that farmers take many more issues than just profit into consideration when making decisions about whether or not to adopt new agricultural technologies (Vanclay, 1992; Willock *et al.*, 1999). For example, Vanclay (2004) describes 24 non-profit considerations that farmers have when making decisions about their agricultural practices, including time constraints, family traditions, sustainable practices and perceptions of risk. Additionally, Edward-Jones (2006) cites five characteristics that significantly impact farmer behaviour: socio-demographics; psychological make-up; characteristics of the farm household; structure of the farm business; and wider social milieu.

However, these criticisms do not negate the contribution of neoclassical methods to our understanding of how attractive agricultural technologies are to farmers or to where they fit within the broader process of sustainable development. For example, Grubb (2014) highlights neoclassical and welfare economic approaches as one of three fields of theory, all of which must be harnessed in order to address the so-called "super wicked problem" (Rittel and Webber, 1973) of climate change

(Lazarus, 2008). The first of Grubb's theory domains is behavioural economics, which considers how and why individuals behave under certain circumstances. This field is where most of the critique for assumptions of farmers as rational economic agents comes from. The second domain is that of neoclassical and welfare economics, which Grubb sees as the "workhorse framework" for most economic analysis, which has informed our understanding of business cycles and financial phenomena such as trade and investment flows. The third and final domain is that of evolutonary and institutional economics, which explores the longer-term development of economic systems and their relation to institutional frameworks. Grubb (2014) argues that consideration of each of these domains is essential and specifically addresses critiques of the neoclassical domain by stating that:

" ... the insights of neoclassical economics do not all hinge on its core assumption of "rational representative agents" as a precise description of reality. Its conclusions remain relevant for as long as that is a better assumption than other alternatives." (pg. 56)

Moreover, although critiques of neoclassical economic theory as a guide to farmer decision making emphasise the many issues other than profit-maximisation that farmers consider, they do also recognise that econonmics is a consideration for farmers. In many ways, it can be assumed that a technology being profitable is a necessary, though not sufficient, criteria for agricultural technology adoption, particularly if biochar is to be adopted by many farmers on a global scale. As such, neoclassical economic theory and its associated methods are used throughout this thesis, on the understanding that they provide an important, though not complete, lense through which to view the validity of win-win claims for biochar.

The following sections provide explanations for each of the neoclassical economic methods used in this thesis, and Figure 2.1 provides a schematic overview of the thesis structure, chapter contents and where each of the methods is used.

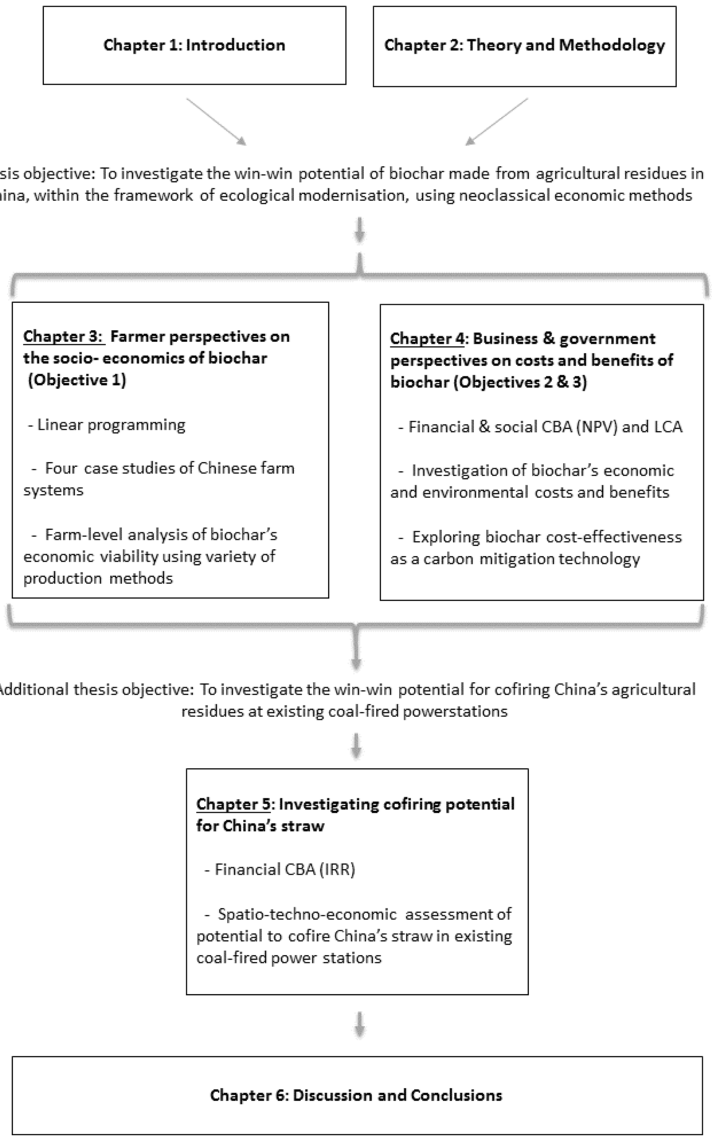


Figure 2.1: Schematic diagram of the thesis structure

2.2 Cost-benefit analysis: an overview

Under neoclassical economic theory, an individual or institution is assumed to take actions that will maximise their profits. It therefore follows that these individuals/institutions must perform some sort of cost-benefit analysis (CBA) in order to weigh up the benefits that they expect to accrue relative to the costs that they expect to outlay, and this concept of cost-benefit analysis is used widely to inform understanding of individual decision-making, technology assessment and policy development, amongst other things.

As a method, cost-benefit analysis has been used extensively by governments, academics and NGOs. It takes much of its theoretical base from the welfare theory of economics, which studies the predicted and/or realised aggregation of benefits to individuals/groups following a proposed intervention. CBA was originally used for initiatives such as water resource development and dam construction (Hanley *et al.*, 1993; Dryzek, 2013), however it has since branched out to assess topics such as wildlife preservation, air pollution and human health, amongst others (Freeman, 1982; Boyle and Bishop, 1987; Johannesson and Jönsson, 1991). Overall CBA is a versatile tool that can easily be adapted to different project situations.

The objectives of this thesis cover the perspective of a range of stakeholders, including farmers, businesses and national/international institutions with an interest in the success of carbon sequestration technologies. The following sections therefore outline the different approaches to cost-benefit analyses that are used to explore each of these differing stakeholder perspectives.

2.2.1 The farmer perspective: linear programming

Linear programming (LP) is a quantitative mathematical analysis tool, which has been developed to support decision-making under complex circumstances where multiple different resources must be allocated between a range of competing uses, often with innumerable possible end point combinations. It has been widely used in the agricultural and farm management sector for tasks such as minimising the cost of nutrient provision for dairy herds (Dent *et al.*, 1986), modelling farm adaptation to climate change (Gibbons and Ramsden, 2008) and cropping/farm management decisions on small-scale farms in developing countries (Lee *et al.*, 1995; Siegel and Alwang, 2005).

Typically LP models are created by listing the resources available, the various activities that require part or all of those resources, and the feasible limits of the system. Constraints are built into the model to ensure that all outcomes are feasible within the real life system being modelled and follow the farm manager's goals, i.e., constraining the amount of land available, or allocating a minimum area to a given crop regardless of its profitability. Each activity is then assigned a value, often monetary, and an objective function (usually net farm profit) is then maximized or minimized (depending on the analysis goals) by increasing or decreasing the levels of each activity. The standard LP equation is:

maximise $c^T x$

subject to:

$Ax \leq b$

$x \geq 0$

where $c^T x$ is the objective function to be maximised or minimised, consisting of c as a vector of coefficients, $(.)^T$ as the transposed matrix, and x representing the vector of variables. The inequalities of $Ax \leq b$ and $x \geq 0$ are constraints, which represent the space within which the objective function is to be optimised, where c and b are vectors of coefficients and A is a matrix of coefficients.

The advantage of LP over other economic tools, such as financial cost-benefit analysis, is its ability to consider a wide range of values for each activity (Bender and McCarl, 1992). For example, a model can consider a range of fertilizer costs, grain sale prices, and biochar impact levels, enabling the comparison of multiple hypothetical scenarios and facilitating our understanding of their influence on overall farm economics. This is particularly important for a technology like biochar, where both the types and scale of impact can be highly variable.

Linear programming was therefore used to analyse the financial viability of biochar within the farm-systems data collected from China. This quantitative economic analysis of biochar's potential in these systems is used in Chapter 3, alongside contextual data from interviews, to appraise biochar's potential as an attractive agricultural input and poverty alleviation technology. Specifically LP is used to explore what combination of grain yield increases (% increase over baseline) and chemical fertiliser application reductions (% decrease over baseline) would produce sufficient profit to break-even on farm-level investment in biochar production/purchase and application. This approach represents a significant departure from the biochar literature that was available in late 2011, which focused predominantly on reporting yield increases from biochar field trials, but not considering whether they were economically sufficient to justify the costs biochar addition for farmers. Finally, in addition to determining whether biochar is a profitable technology for farmers, LP was also used to explore what necessary yield increases and fertiliser reductions would need to result from biochar application, under a variety of economic scenarios, in order for farmers to break-even on their investment.

2.2.2 The business perspective: financial cost-benefit analysis

Having used farm-level linear programming analyses for Chapter 3, the next logical step in understanding biochar's potential in China is to examine it from the perspective of businesses. For businesses it is natural to assume that bottom-line profit is a key driver of investment behaviour, and therefore financial cost-benefit analysis is the obvious methodological choice for situations where there are a discrete number of investment options to be compared.

In comparison to LP, financial CBA is suited to economic situations where the costs and benefits of a discrete number of options need to be compared, particularly if these options involve financial flows that occur over a period of time, usually years. This is due to the "time value of money", which acknowledges that money that is available to an individual today, i.e., \$1, has a greater value than the same \$1 at a future point in time. Therefore, when comparing the present-day value investment projects that involve different inflows and outflows of cash over varying time periods, it is important to take the time value of money into account by converting the value of all future cash flows into a single present value. Subtracting the current value of all cash outflows from the current value of all cash inflows calculates the Net Present Value (NPV). Often this is used as an indicator of whether a specific project will generate a profit (i.e., the NPV is positive) and also to compare the profitability of projects (i.e., the project with the highest NPV should generate the greatest profit.) The standard equation for calculating

NPV is:

$$\text{NPV} = \sum_{t=1}^{T} \frac{C_t}{(1+r)^t} - C_0$$

where C_t is the net cash inflow during the time period, C_0 is the initial investment, r is the discount rate, and t is the number of time periods.

The discount rate in this equation is a key feature of calculating the NPV, as this is the rate at which future cash flows are adjusted to the value of present day cash flows. Setting an appropriate discount rate can be difficult, as it depends on issues such as inflation, investment risk and the "cost of capital" (i.e., how much it costs to use money, either in terms of borrowing it from others or by spending your own money and not earning interest on it). Although there is no universally accepted method for choosing a discount rate, it is common to use either the opportunity cost of capital (i.e., the return that would be received if the funds were invested in the private sector) or alternatively a national interest rate, i.e., the cost to the national government of borrowing.

An alternative to calculating the NPV of a project is to calculate the internal rate of return (IRR), i.e., the expected rate of return that an investor can expect on their investment over a specified time period. Alternatively it can be thought of as the discount rate that renders the NPV of a project as zero. Although IRR is a useful indication of the return that an investor might receive, it tends not to be used to compare mutually exclusive project options. NPV is the preferred valuation tool for comparing projects, because a project with a higher investment may have a lower IRR, but a higher NPV (i.e., a greater monetary value to an investor). Therefore IRR tends to be used for standalone investment decisions, rather than comparisons.

For these reasons, NPV was the cost-benefit indicator used to compare the economic benefits of turning China's agricultural straw into biochar as compared to either electrical or heat energy (Chapter 4; Objective 2), whereas IRR was the cost-benefit indicator used to investigate the financial viability of individual coal-fired powerstations choosing whether or not to cofire straw residues with coal (Chapter 5; Objective 4).

2.2.3 The climate change mitigation perspective: social cost-benefit analysis

If biochar is to contribute significantly to climate change mitigation, it will need to be produced and used by a large number of actors on a global scale. In order to do this, it must be an economically rational investment option for multiple actors, from farmers to businesses. However early cost-benefit analyses of biochar's economic potential suggested that it would not be financially viable without some form of subsidy (Brown *et al.*, 2010; Roberts *et al.*, 2010). Given biochar's "win" potential as a carbon mitigation technology, one approach is to place a value on the carbon sequestration services that biochar provides, which are typically not valued in the market-place. Placing a value on a societal good, such as carbon sequestration, is a form of social cost-benefit analysis, in which costs and benefits bourne by or afforded to society are included in the cost-benefit assessment (Jones *et al.*, 1990). However, if biochar is to be subsidised by governments and/or environmental actors on these grounds, it will need to demonstrate value for money in this regard: the cost of carbon mitigation using biochar as the technology of interest would need to be less than or equal to the cost of other technologies providing comparable carbon mitigation services. In short, biochar must be a cost-effective climate mitigation strategy in comparison to other options (Chapter 4, Objective 3).

One approach is to determine whether biochar is a cost-effective climate change mitigation option is to calculate the cost per tonne of carbon dioxide equivalent sequestered that must be provided by investors in order that a given biochar project breaks-even and/or generates a specified return. This requires investigation of both economic and environmental costs and benefits for a given biochar system and comparator systems. The economic component of this analysis was therefore conducted using the CBA method described above, and the environmental outcomes were analysed using life-cycle analysis, described in the following section.

2.2.4 The climate change mitigation perspective: life-cycle analysis

In order to understand the full range of biochar's environmental impacts, it is crucial to investigate the entire biochar production system (Scholz *et al.*, 2014). Variations in the biomass feedstocks, their pre-existing uses, biochar production technologies, and the agro-ecosystems within which biochar is applied can all have significant impacts on whether the net impact of biochar is positive or negative according to a chosen environmental indicator.

Life cycle analysis (LCA) is a method often used to assess the environmental impacts associated with a given product or technology, and is typically described either as attributional/descriptive (where the environmental burdens associated with a defined product or service are calculated for a specified point in time, typically the recent past) or as consequential/predictive (where the consequences of change(s) to an existing system are considered, taking into account any knock-on systemic impacts on e.g., economies or markets). LCA is an increasingly popular tool with which to assess the environmental impacts of biochar systems, and the biochar LCA studies published to date have typically been calculated using the attributive approach (Roberts *et al.*, 2010; Shackley *et al.*, 2011b; Lugato *et al.*, 2013; Scholz *et al.*, 2014). LCA is a powerful tool for analysing environmental impacts due to its flexibility, whole systems approach and the transparency afforded by the ISO 14040 standardised methodology (ISO, 2006). Moroever, LCA can easily be combined with cost-benefit analyses such that the lifetime, systems-level environmental and economic impacts of a given biochar intervention can be assessed together.

Typically, a LCA is made up of four stages:

1. Defining the goal and scope: In this stage the indicator(s) of interest are defined (for example, climate change) and the functional unit of the analysis should also be specified (for example GHG emissions per tonne of feedstock, per household, or per unit energy).

2. Inventory analysis: In this stage, data availability is assessed. Data may come from primary experiments, interviews, and/or secondary data sources (such as published reports, academic papers and websites.) Quite often, data is limited and not at the level of specificity that would ideally be available. In these circumstances, it is common for LCA practitioners to generalize from data collected in other areas, or at other times. However, the assumptions made when drawing these generalisations must be made very clear.

3. Impact assessment: Once the data is collated and allocated to each process

within the system, the environmental impact of a given technology can be assessed. For example, in Chapter 4 the flows of CO_2, CH_4, and N_2O for various biomass pathways are summed and the baseline flows subtracted from them. This gives an indication of the relative GHG impact for different biomass processing options.

4. Interpretation: Following the impact assessment stage, the results must be interpreted in the larger context of the system and individual parameters tested for their individual contributions to the overall outcome. This process, known as sensitivity analysis, is carried out by varying each parameter value independently and assessing its impact on the overall system result. This process ensures that the sensitivity of the overall system to changes in individual parameter values is understood, which is a much more meaningful knowledge contribution than simply reporting a single system value. Given the importance of this approach to LCA validity, sensitivity analysis was employed for all LCAs and CBAs conducted in this thesis.

A combination of LCA and social-CBA analyses were used in Chapter 4 (Objective 3) to compare the cost-effectiveness of biochar as a climate change mitigation strategy with the dominant existing uses of China's straw feedstocks, under a range of public policy support scenarios.

2.3 Data Collection

A combination of primary and secondary data were used to build the various cost-benefit and life-cycle analysis models outlined in the previous sections. The following sub-sections describe the data sources, how the data was collected, and the rationale for the methods used.

2.3.1 Primary data collection: semi-structured interviews

It could be argued that the data required to produce linear programming and cost-benefit analyses on agricultural systems in China is available through desk-based research, thus negating the need for in-depth fieldwork. Indeed, some models for biochar development and adoption have been built using such data (Woolf *et al.*, 2010b). However, it can also be argued that personal experience of agricultural systems and first-hand discussions with actors within those systems is necessary in order to get a deep and nuanced understanding of whether a new agricultural technology like biochar is appropriate.

Moreover, there has historically been a tendency for the development of new agricultural technologies, and their transfer to farmers, to be done in a top-down manner. For example, much has been written about the evolution of agricultural extension programmes, which began as official Government programmes in many countries shortly after World War II, when the agricultural sector of many nations required a kick-start to re-boost their productivity levels (Birkhaeuser *et al.*, 1991). The main assumption was that technologies that were developed in scientific laboratories could be transferred to farmers through a linear process of research, followed by activities to raise farmer knowledge about the technology, leading to the transfer, adoption and eventual wide-spread diffusion of these technologies over time (Ison and Russell, 2007). This became known as the Transfer of Technology (TOT) model (Biggs, 1990).

This model of agricultural extension soon spread into aid programmes globally, receiving significant funding from the World Bank, and instigating the creation of networks of agencies for agricultural research, both nationally through National

Agricultural Research Systems (NARS), and internationally with the Consultative Group on International Agricultural Research (CGIAR) and the International Rice Research Institute (IRRI) (Biggs, 1990). The aim was for CGIAR and IRRI to develop generalised agricultural solutions, and for the NARS to contextualise them to their national and local situations via agricultural extension workers who visited farms to provide information and/or training (Scoones, 2009).

However, despite huge financial and human resources being invested into agricultural extension programmes, their success has been mixed at best. This may in part be due to a lack of rigorous data collection with which to evaluate the programmes (Birkhaeuser *et al.*, 1991). However, others claim that agricultural extension programmes were not accompanied by sufficient supportive structures, such as transport infrastructure, credit availability, natural resources and/or markets through which to sell higher value goods. Where farmers could not see a way to sell their increased produce, or could not access sufficient credit to invest in a new technology, then their motivation for and potential benefits from technological adoption was thought to be hindered (Röling and Kaimowitz, 1990).

Another explanation is also that the rigid, top-down structure of the TOT model means that farmers are unable to take ownership of, or influence, the technologies that are being developed and extended to them (Biggs, 1990). This lack of interaction between research and farmers has arguably led to a “fix” mentality, where new technologies are developed by researchers distant from the local situation, whose goals are very different from those of local people (Ison and Russell, 2007), and where the one-way flow of information from research centres to local farmers ignores the farmers’ own innovation capacity, local knowledge and decision-making skills. This might explain why technology transfer programmes tend to work best where recipients undertake their own on-farm R&D projects, such as in later extension programmes like the Training & Visit model, and Farmer Field Schools (Scoones, 2009).

With this in mind, it was felt that primary data collection was essential for this PhD to meet its objective of providing a data-informed, applied perspective on the socio-economic suitability of biochar to farmers across a range of Chinese farming systems. Moreover, at the start of the PhD in 2011 there was no available research on the socio-economic fit of biochar in existing agricultural systems outside the western world, either using or not using primary data obtained through first-hand fieldwork experience.

Finally, taking a case study approach to the initial data collection procedure was also felt to be warranted. The following sub-section provides a brief discussion on the rationale for case study-based research, before providing a justification for the selection of the field sites.

2.3.2 Rationale for the case study approach and selection of sites

Case study research can be criticised for its perceived inability to contribute to broader-scale theoretical advancement of a given research area. The difficulties in generalising findings from one case study location to another, means that this method may not produce research findings that can be used to prove high-level hypotheses or generate systematic unbiased data (Flyvbjerg, 2006).

However, at the beginning of this PhD in 2011, it can also be argued that case studies were exactly what the social science literature on biochar systems needed, in order to provide some local context to the global and regional scale models for biochar use and carbon sequestration that were being developed. At that early stage in the process of building understanding of appropriate biochar systems, case studies could provide much-needed contextual information about how biochar might fit within existing agricultural systems, and which features were most critical to that fit. Indeed, there were already some case study economic analyses of biochar systems available in the literature in 2011, however they were all based in developed country and commercial settings. There were no analyses performed from the socio-economic perspective of smallholder farmers, and therefore it was felt that local case study data collection and analysis could provide a new contribution to the field of biochar research.

Fieldwork to collect case study data was therefore carried out over 6 months in the summer of 2012, during which time over 150 interviews were carried out across contrasting farming systems in four Chinese provinces: Heilongjiang, Henan, Jiangsu and Yunnan.

Sites were selected on the basis of the presence of biochar field trials and/or demonstration pyrolysis units; in order to compare and contrast a diverse range of agricultural systems; and on ease of access via University and government contacts. Researchers spent up to three weeks living with local families at each site, undertaking semi-structured interviews on farming practices, inputs, outputs and socio-demographic characteristics with village leaders, farmers and businesses. Table 2.1 provides an overview of the characteristics of the case study sites, with further information provided in Chapter 3.

Table 2.1: Overview of case-study site characteristics

Province	Description	Main Crops	Average Farm Size (ha)	No. of Interviewees
Heilongjiang	North-eastern, one growing season, state-run farms, mechanised	Maize, rice, beans	25	15
Henan	Central, agriculturally-dependent, large out-migration	Wheat, maize, peppers	0.46	56
Jiangsu	Eastern, industrialising, plentiful off-farm work	Rice, wheat, vegetables	0.12	33
Yunnan	South-western, tropical, mountainous, low-income	Rice, maize, wheat, sugarcane, tobacco, walnuts	0.65	45

2.3.3 Semi-structured interviews and the Sustainable Livelihoods Approach

Interviews were used at each case study location to investigate the socio-economic characteristics of the various farming systems. All interviews were semi-structured, ensuring that similar issues were discussed and data was collected across a consistent set of topics, but allowing the interviewees to direct the interview process to areas that they felt were important and/or not covered by the pre-prepared questionnaire. All interviews were started with an introduction about why the research team was there, the purpose of the project, and a standard verbal ethics procedure to ensure informed consent from each interviewee before proceeding. Interviewees were also assured that they could leave the interview at any point, and were not required to answer any questions if they did not want to.

The list of topics that made up the structured part of each interview was guided by the Sustainable Livelihoods Approach (SLA). The SLA was developed in the early 1990s, with strong links to academics working in the field of participatory development research (Chambers and Conway, 1992; Scoones, 1998). The core aim of the SLA is to understand how different people in different places live according to their means of gaining a living, the resources that they have available, and their subsequent chosen economic activities. SLA pulled together theories and practice from Rapid Rural Appraisal and Participatory Rural Appraisal alongside household economics, gender analyses, political ecology and resilience studies, and became a popular approach for the UK's Department for International Development, who adopted it as their standard tool for development projects overseas (Scoones, 2009).

Over many years of use, the SLA has been criticised for its static structure, narrow focus on local issues, inability to link these issues to the wider macro-economic picture, and a lack of integration with long-term issues such as climate change, environmental degradation and economic shifts (Scoones, 2009). However, it is arguably still a helpful starting-point from which to investigate the livelihood strategies of rural communities in developing countries, and a solid foundation around which to design a questionnaire on the decision-making environment of smallholder farmers. An example questionnaire from the surveys conducted across China is available in Appendix 1.

2.3.4 Sampling strategy

The objective of the thesis was to provide an overview of the socio-economic potential of biochar within a range of agricultural systems. Similarly, within each case study site, every attempt was made to capture the diversity of agricultural and livelihood strategies that were in evidence.

Unfortunately, comprehensive lists of inhabitants at each case study site were either unavailable or nonsensical when provided. Therefore a stratified sampling approach was employed to select interviewees, based on socio-demographic factors such as income (proxied by the visual appearance of the house), geographic location (ensuring a wide geographic spread of interviews within the village area), and interviewee age. Interviews in any given place were considered complete once data saturation had been reached, i.e., interviewers were consistently returning with information similar to that already collected, despite efforts to find fresh or different stories. The result is therefore not a proportional sampling of each village's inhabitants, but instead aimed to capture the variety of individuals and livelihood strategies present. With this information, it is anticipated that there is sufficient data to understand firstly how the agricultural systems vary, and secondly to what extent these differences affect the adoption potential of biochar.

Questionnaires were delivered face to face, approximately half with both the translator and myself there (often the earliest interviews in a new location) and half with just the translator. My presence in the interviews was often beneficial for probing deeper into qualitative questions or following interesting leads that surfaced during interviews. However this advantage may have been counter-weighted by the increased reticence of some interviewees to talk openly in the presence of a foreigner. Therefore I was present in the first interviews in each village, to set up the interview frame and ensure relevant questions were being asked. Later interviews could then be done by a lone translator, ensuring minimal disruptive influence from my presence.

The data taken from these interviews forms the basis for much of the analyses contained within this PhD, particularly the linear programming models in Chapter 3, and to a lesser extent for the cost-benefit analyses in Chapter 4. Chapter 5 consists entirely of secondary data collected from partner organisations in China and through online searches. Detailed information on that data collection process is provided within the text of the chapter.

Chapter 3

From rhetoric to reality: farmer perspectives on the economic potential of biochar in China

This chapter has been published as:

Clare, A., Barnes, A., McDonagh, J., & Shackley, S. (2014). *From rhetoric to reality: farmer perspectives on the economic potential of biochar in China.* International Journal of Agricultural Sustainability, 12(4), 440-458.

Author Contributions

Abbie Clare: Formulated the concept for the paper, designed questionnaires, collected data in China, cleaned data, constructed linear programming models, wrote paper

Andrew Barnes: Provided guidance on the theory and application of linear programming
John McDonagh: Provided funding for a second round of data collection through the Sustainable Agriculture Innovation Network and contributed to discussions on questionnaire design
Simon Shackley: Provided overall guidance on paper concepts, and gave suggestions for the final draft of the manuscript

3.1 Chapter Rationale

This chapter addresses thesis Objective 1: to investigate whether biochar's agronomic benefits (defined as changes to net-farm profit from increased crop yields and/or reduced fertiliser use resulting from biochar application) can outweigh the costs of biochar sourcing and application (definfed as $ per kg biochar produced and applied) for Chinese farmers across a variety of farming systems. This is done with a view to assessing both the likelihood that biochar might be adopted by farmers in China, and also to assess its poverty alleviation potential

3.2 Introduction

Biochar is theoretically a promising technology for the challenges facing Chinese agriculture. Firstly, China is renowned for its high population to arable land ratio, housing 22% of the global population with only 9% of the world's arable land. Thus any technology that can improve agricultural productivity is potentially of interest to the government, with preliminary evidence from Chinese field trials suggesting that biochar can stimulate crop yield improvements (Zhang *et al.*, 2010a, 2012a; Wang *et al.*, 2012a). Moreover China has a very large supply of possible biochar feedstocks, as a result of the widespread burning of agricultural straw, which has become standard practice for many of China's 200 million small-scale farmers (National Bureau of Statistics of China, 2003; Sun and Sun, 2006), releasing an estimated 107 million tonnes of CO_2e into the atmosphere each year (Zhao *et al.*, 2011).

For biochar to be adopted on a large-scale, it will need to be socio-economically suitable to farmers in China's agricultural systems. However very little is known about the economics of production and application of biochar in real world systems. Although some cost-benefit analyses exist, these are predominantly based in developed countries such as the US or UK (Brown *et al.*, 2010; Roberts *et al.*, 2010; Shackley *et al.*, 2011b), and focus on medium/large-scale profit-making enterprises rather than the decision-making environment of smallholders.

This paper therefore provides an overview of four contrasting agricultural systems in China, their social and economic structures, and investigates which biochar production technologies may or may not be appropriate and/or desirable for Chinese farmers. In each case, we investigate the level of yield increases and fertilizer use efficiency improvements that biochar must facilitate in order to break-even with the costs of its production/purchase and application to soil. We also consider whether these yield increases and fertilizer use decreases are feasible, according to published data on biochar's agronomic impacts. Finally we compare the additional farm-revenues that biochar is capable of generating to other livelihood options available to farmers, in order to assess the potential of biochar as a poverty alleviation strategy.

3.3 Methodology

3.3.1 Site selection and data collection

Uptake of agricultural technologies often depends on local characteristics of the community in question. In the absence of verifiable local data sets, primary data collection was deemed necessary to ensure high data quality and site-specific relevance. Therefore four Chinese villages and towns were chosen in which to conduct face-to-face household surveys. These sites were spread across four geographically-, climatically- and socially-contrasting Chinese provinces: Heilongjiang, Henan, Jiangsu and Yunnan (Figure 3.1). Climate and soils data are taken from Hu and Zhang (2006) and crop data is taken from surveys conducted as part of this research project.

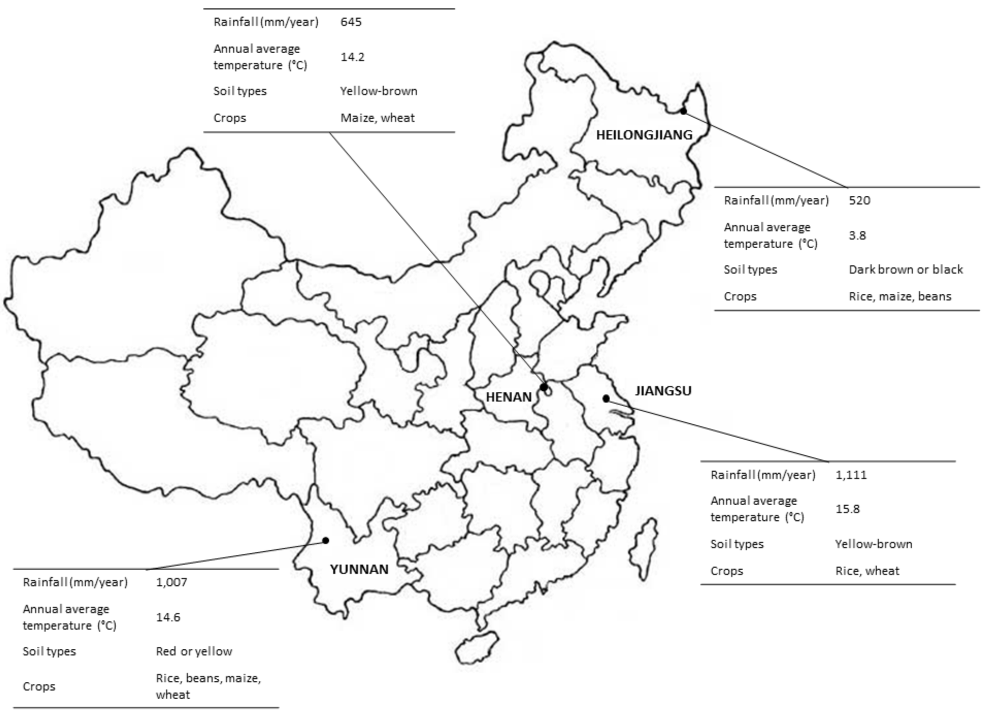


Figure 3.1: Case study sites and provincial information

Sites were selected on the basis of the presence of biochar field trials and/or demonstration pyrolysis units; in order to compare and contrast a diverse range of agricultural systems; and on ease of access via University and government contacts. Researchers spent up to three weeks living with local families at each site, undertaking semi-structured interviews on farming practices, inputs, outputs and socio-demographic characteristics with village leaders, farmers and businesses. For household surveys, a purposive sampling strategy was used to maximise the diversity of respondents, covering the variety of available livelihood options and

Table 3.1: Socio-demographic & agricultural indicators in Henan, Heilongjiang, Jiangsu and Yunnan provinces

	Heilongjiang	Henan	Jiangsu	Yunnan
Total population (millions)	38.3	94	78.7	46
Life expectancy (years)	66.97	70.15	71.37	63.49
Rural per capita net income (USD year $^{-1}$)	995	885	1460	630
Total farmland (hectares)	11,830,100	7,926,400	4,763,800	6,072,100
Land area mechanically cultivated (%)	89.10	86.40	82	9.76

living conditions. Participants were therefore chosen according to indicators of income and age, as well as on- and off-farm work.

3.3.2 Overview of the four case study provinces

Table 3.1 outlines each provinces key agricultural and demographic characteristics, according to government statistics (National Bureau of Statistics of China, 2008, 2013).

Jiangsu province is eastern, industrialising, and factories are fast becoming the primary source of income for families. It therefore has the wealthiest rural population (as defined by per capita net income of rural households) of these case studies, closely followed by Heilongjiang (where there are many large-scale, state-managed farms), and Henan (where there are fewer factories but working-age adults tend to out-migrate and send money home). The poorest case study site is Yunnan, where mountainous topography, degraded soils and an agriculturally dependent population lead to minimal mechanisation and low rural incomes.

The following four sub-sections provide more detail on the case studies. Unless otherwise stated, this descriptive information comes from the interviews conducted during this project.

Heilongjiang province: 920 farm

In addition to small household farming systems, China also has large state-managed farms, mostly located in the northern provinces of Heilongjiang, Xinjiang, and Inner Mongolia. These are vast areas of fertile land, portions of which are rented out to tenant farmers via government-run farm bureaus.

The case study in Heilongjiang is based on 920 farm, Suibin county. Heilongjiang is known as the great northern granary, containing one twelfth of China's cropland and one sixth of its commercial grain production area (Muldavin, 1997). Farmers renting land from the 920 farm bureau work an average of 25 hectares (ha), planting rice, maize or beans during the summer season, and leaving the land fallow during the frozen winter months. Although farmers planting each crop were interviewed, only those planting rice are considered for analysis due to the availability of rice agri-residue biomass for biochar production. Beans produce very little non-crop biomass, and maize straws and cobs are chopped and returned to the soil by harvest machines, rendering them unavailable for collection. By contrast, rice straw is burned in the field, thus providing the most suitable waste product for conversion to biochar. Rice land and beans/maize land are also situated far from each other; therefore it is not practical for biochar made from rice plant residues to be applied to maize/beans.

Henan province: Kong and Dou Lou villages

Kong and Dou Lou villages sit within Kongzhuang township, Xiayi county. They have a relatively arid climate, where summer maize and winter wheat are the main crops, alongside a small proportion of land given to vegetables such as peppers. These villages were chosen due to their proximity to a large-scale biochar producing factory (Sanli New Energy Company), in order to discuss farmer experiences with biochar and investigate the process of straw collection and pyrolysis being undertaken by Sanli.

Henan is often described as agriculturally dependent (Wang *et al.*, 2012c), however it is also being influenced by China's industrialisation, as many of its working-age adults migrate to other provinces in search of jobs. Rural incomes have continued to rise as a result of this migration, with the proportion of income from agriculture decreasing from 66.3% in 1985, to 29.1% in 2010 (Huang *et al.*, 2012b). Yet despite farming's diminishing economic importance at the national level, it remains an important source of income and family stability in central provinces such as Henan, and accounts for around 16% of household income.

Jiangsu province: Jing Tang village

Jing Tang village sits within Yixing municipality, and was chosen firstly to represent the industrialising face of China and secondly due to the presence of a multi-season biochar field trial. Farmers predominantly grow grains; rice in summer and wheat in winter. With an ample availability of nearby factory jobs, few households maximise on-farm profit through diversifying crops into higher margin vegetables. As a result, these farmers derive just 6% of their total annual income from farming activities (Table 3.2)[1], preferring to work in local factories paying

[1]One hectare is equivalent to 15mu, which is a Chinese unit of land used by most smallholders to indicate the size of their farm. Farm income does not include costs of labour or machinery and is estimated for rice farmers only.

US$16-32[2] day^{-1}.

Table 3.2: Socio-demographic averages for each case study

	Heilongjiang	Henan	Jiangsu	Yunnan
Interviewees	15	56	33	15
Age	46	53	64	49
Farm size (ha)	25	0.46	0.21	0.65
Off-farm income (USD year^{-1})	0	3,970	3,565	505
Farm income (USD year^{-1})	48,720	750	245	1,535
Farm income (as % of total)	100%	16%	6%	75%
Non-grain land (as % of total)	0	5	0	34

Yet despite the low economic importance of agriculture, most households continue farming rather than rent out their land. Farmers reported many reasons for this, including uncertainty about how renting would affect land rights (Ding, 2008); the high compensation rates available if a factory requests to build on their land; and concerns over food safety. However the result is perpetuation of small plots managed by time-constrained households. With little time to spare, relatively high incomes, and farming subsidies received from the government (Gale *et al.*, 2005), these farmers are over-applying fertilizer and pesticides with significant negative impacts on soil, water and human health (Huang *et al.*, 2000; Zhang *et al.*, 2013a).

Yunnan province: Ping Zhang village

Ping Zhang village is part of Yangliu township, Longyang district. It sits 2000m

[2] One USD is equivalent to 6.25 Chinese yuan

above sea level in China's south western mountains, with a consistently mild climate throughout the year. Ping Zhang is a 90minute motorcycle ride from the nearest city, Baoshan, and is therefore far from off-farm sources of income, such as factory work. Families are generally self-sufficient in rice, and cannot sell surpluses due to the absence of a rice market. Wheat and maize are mostly fed to pigs, which are then sold. A small proportion of beans are eaten, but the majority are sold for cash income.

Of the four case studies, Ping Zhang is the poorest farming community. In recent years government initiatives have encouraged cash crops such as tea, tobacco and walnuts, however agricultural advancement is held back by the lack of mechanisation potential and the isolation of communities from markets.

3.4 Data analysis

A set of single-objective, single-period, deterministic linear programming (LP) models were built for each case study site, based on data collected during interviews. Specifically, these models represented the flow of inputs and outputs within each site at a typical farm level (Table 3.3[3]).

The baseline model for each site was created solely from the quantitative interview data, in order to create the input-output coefficients, the level and direction of

[3]Yunnan fertilizer prices are uniform across crops because farmers were unable to identify how they divided fertilizer between their various land portions and crop systems. They were only able to report the total spent on fertilizer each year. For the purpose of the model the total value of fertilizer was therefore divided equally according to land size in order to generate an average cost of fertilizer per ha.

Table 3.3: Key model parameter values derived from local data

	Heilongjiang	Henan	Jiangsu	Yunnan
Labour (USD hour^{-1})	2.8	1.9	2	1.2
Rice yield (t ha^{-1})	8.6	-	7.6	5.8
Maize yield (t ha^{-1})	-	5.5	-	7.1
Bean yield (t ha^{-1})	-	-	-	3.8
Wheat yield (t ha^{-1})	-	5.4	5.4	2.9
Rice fertilizer (USD ha^{-1})	457	-	647	185
Maize fertilizer (USD ha^{-1})	331	401	-	185
Bean fertilizer (USD ha^{-1})	685	-	-	185
Wheat fertilizer (USDha^{-1})	-	438	428	185
Rice (USD kg^{-1})	0.41	-	0.45	0.77
Maize (USD kg^{-1})	0.22	0.31	-	0.32

each constraint, and adequately represent the number of available activity options and localised prices. Exploratory data analysis found no significant relationships between grain yield/straw use and demographic indicators, therefore average self-reported grain yields were used to estimate productivity from the survey data collected. Average straw biomass availability was calculated using published conversion product-residue ratios of grain:straw (Zhang and Zhu, 1990) and reported household straw uses. Finally the financial value of farming activities was modelled according to local conditions. For example, grain sale price, labour cost, and fertilizer expenses were calculated by averaging interview-derived values. Table 3 details the variations in these parameters across sites.

Fertilizer application was highest in Jiangsu, followed by Henan, Heilongjiang, and then Yunnan. Thus, the highest rates were in small farms of relatively wealthy households. There is as yet no consensus on the effect of farm size on productivity or fertilizer application rates (Fan and Chan-Kang, 2005), however the pattern observed here is certainly explicable within the context of China's agricultural landscape, where it appears that time-constrained, small-scale, relatively wealthy farmers in Jiangsu apply high rates of fertilizer compared to those in Heilongjiang's large, mechanised farms, where farmers have contact with extension services and invest all their working time in maximising their farm profit. Finally, it is also logical that the low-income and isolated Yunnan case study had the lowest rates of fertilizer use, due both to difficulties in affording and accessing fertilizer products.

Overall reported fertilizer use was high compared to other published estimates. For example Zhang *et al.* (2013c) conducted a survey across 2,346 villages in 27 provinces of China, reporting an average N application rate of 209kg ha^{-1} for rice, whereas the average in Jiangsu was 302kg N ha^{-1}. Similarly N application for wheat in Henan and Jiangsu were 224kgN ha^{-1} and 277kgN ha^{-1} respectively, whereas Zhang *et al.* (2013c) report 197kgN ha^{-1}. However, it should be noted that Zhang *et al.* (2013c) also have extremely large confidence intervals around these estimates, of +/- 140kgN ha^{-1} for rice and +/-134kgN ha^{-1} for wheat, indicating a wide variability of fertilizer practices in China. If farmers in our sample systematically over-reported their fertilizer use, the effect would be for the models to over-estimate economic savings from fertilizer use efficiency following biochar application. However all models were relatively insensitive to changes in fertilizer use, therefore any over-estimation is unlikely to have significantly impacted the results.

Valuations of farm produce were restricted to grains, thus excluding fruits, vegetables and other cash crops. In Heilongjiang, Henan and Jiangsu this is logical, because just 0-5% of land was planted with cash crops (Table 2). In Yunnan an average of 34% of land was given over to non-grains, however these were crops such as walnut trees and tobacco, for which biochar's impact is not well understood. Furthermore, only sparse information was available on the input and output economics for these crops, as they were a relatively new addition to this farming system.

Each baseline model was constructed, calibrated and checked against interview data to ensure realistic representation of the farms. The single objective function was to maximise farm level profits. In total, there were 21 activity options and 15 constraints. For example, Jiangsu's baseline model predicts that all available straw is sold, and this supports our observations of straw selling in Jiangsu, mostly by older generation household members with limited alternative work opportunities. However, younger people did not take time to collect and sell straw, because returns to labour are better in factory jobs. Likewise, the Yunnan baseline model predicts that using maize to rear pigs is more profitable than maize selling, which fits observations that most maize in Yunnan is fed to pigs.

3.4.1 Biochar sourcing options

Once baseline models were calibrated, the various options for sourcing biochar were included and compared to the profitability of other biomass uses. Biochar can be produced in farm-, community-, or commercial-scale technologies, with the suitability of each option varying according to the situation. This study parameterised three technology options: a small on-farm kiln, a medium-size on-farm kiln, and commercially purchased biochar.[4] Table 3.4 details these technologies.

Table 3.4: The three options parameterised for sourcing biochar

Scenario	Cost (USD)	Lifetime (Years)	Size (kg straw)	Production temp. (deg. Celsius)	Char yield (%)
Small kiln	32 / kiln	5	10	400-500	33
Medium kiln	400 / kiln	5	500	400-500	33
Biochar purchase	0.26 / kg	-	-	-	-

The small kiln is a design currently being piloted in sub-Saharan Africa, which can pyrolyse around 10kg of feedstock per run. It is intended for use by individual households and is relatively cheap to purchase (US$32). Small kilns tend to be labour intensive, but offer a more accessible entry into biochar production for poorer households. By contrast the medium kiln can pyrolyse 500kg of feedstock per run and is therefore less labour intensive than the small kiln, but has a much higher capital cost of US$400. Finally, large-scale pyrolysis units exist in some provinces, selling biochar for US$0.26kg^{-1}.

Not all technologies were tested for each case study. For example, the small kiln was omitted from Heilongjiang analyses due to the large scale of farming systems operating in this province. Similarly the medium kiln was not included in Yunnan analyses due to the mountainous topography rendering transport between farms infeasible. This created a total of ten case study-technology combinations to be modelled (Table 3.5).

[4] Although work is being done with biochar-stoves, these were not parameterised here because none of the case study sites relied predominantly on biomass for cooking energy. Heilongjiang, Jiangsu and Henan used electricity, gas, or coal, whilst households in Yunnan used biogas obtained from pig wastes.

Table 3.5: The ten case study-technology combinations tested

	Heilongjiang	Henan	Jiangsu	Yunnan
Small kiln		X	X	X
Medium kiln	X	X	X	
Commercial biochar	X	X	X	X

The labour requirements for straw collection and biochar spreading were calculated according to estimates provided in interviews at each site (Table 3.6). Units for straw collection and biochar spreading were defined as minutes per kg collected, and were assumed constant across all crops and case studies.

Table 3.6: Labour assumptions for straw collection, biochar production and spreading

Activity	Labour required (minutes kg^{-1})	Justification
Straw collection	0.6	109kg straw collected per hr = 0.01hours kg^{-1} straw collected = 0.6minutes kg^{-1} straw collected
Spreading fertiliser/biochar	1	60kg fertilizer takes one hour to spread = 0.017 hours kg^{-1} = 1.0minutes kg^{-1} spread

These labour estimates were used to approximate the time taken to produce and spread one kg of biochar for each of the three scenarios outlined in Table 3.4. Table 3.7 displays these calculations. We assume 3kg of straw makes 1kg of biochar, based on personal communication with the small and medium kiln designers.

Table 3.7: Labour estimates for the three biochar-sourcing scenarios

Technology	Labour (minutes kg^{-1} biochar)	Justification
Small kiln	20.8	3kg straw collected = 0.03 hours One kiln run takes 1 hour = 0.3hours kg^{-1} biochar Spreading = 0.017 hours kg^{-1} biochar = 0.347 hours total = 20.8minutes kg^{-1} biochar made and spread
Medium kiln	3.7	3kg straw collected = 0.03 hours One kiln run takes 2.5 hours = 0.015 hours kg^{-1} biochar Spreading = 0.017 hours kg^{-1} biochar = 0.062 hours total = 3.7minutes kg^{-1} biochar made and spread
Commercial biochar	1	One kg of biochar spread = 0.017 = 0.017 hours total = 1.0minutes kg^{-1} biochar spread

On-farm profit was optimised for each of the ten scenarios, across a range of possible biochar-induced yield impacts (0-100% increases, changing in 1% increments) and fertilizer use efficiency improvements (0-30% reductions in fertiliser cost, changing in 10% increments). This created a total of 400 model runs for each case study-technology combination. The value of each incremental change was based on the estimated impact of one kg of biochar under a 10 tonnes per hectare (t ha^{-1}) application rate. Each kg of biochar was assumed to have the same absolute effect on yield and fertilizer use efficiency. The break-even point was then calculated as the combination of yield increase and fertilizer use decrease at which biochar paid back its costs of production and application to soil. Sensitivity analysis was conducted on fertilizer cost, labour cost, straw sale price and grain purchase price, to assess the strength of their influence on biochar's break-even point.

Finally, it is important to note that only yield increases and fertilizer use decreases are considered as benefits from biochar application. Other positive externalities such as carbon storage and reduced fertilizer leaching to the environment are not considered, due to doubts over the viability of carbon markets, the absence of an approved carbon market methodology for biochar, and the desire to only include benefits that are immediately salient to farmers.

3.5 Results

3.5.1 The economics of commercial biochar

The simplest biochar scenario tested is purchasing biochar commercially and spreading it on agricultural land in addition to inorganic fertilizer. The costs are therefore the purchase price of biochar (US\$0.26kg^{-1}) and the labour required to spread it (calculated as 1minute kg^{-1} (Table 3.6) and costed using the local rate for labour (Table 3)). Figure 3.2 displays the percentage yield increase required in order to pay back these costs at each site. The dotted line represents the 15% yield increase level which is generally accepted as biochar's average impact across a range of soils, climates and biochar types (Jeffery *et al.*, 2011; Biederman and Harpole, 2013; Liu *et al.*, 2014a) and which is also supported by results from individual field trials in China (Zhang *et al.*, 2010a, 2012a; Wang *et al.*, 2012a).

Where a range of values is displayed, the upper limit represents the yield increase required with zero improvements to fertilizer efficiency, and the lower limit represents the yield increase required where farmers use 30% less fertilizer.

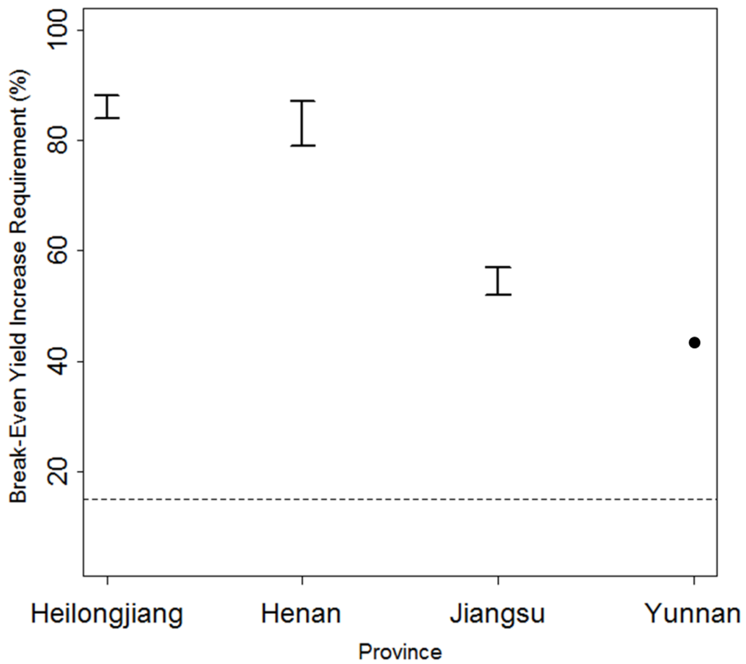


Figure 3.2: Commercial biochar break-even yield increase requirements

The data suggest that, assuming no gains from fertilizer use efficiency, commercial biochar application must increase yields by 43% in Yunnan, 57% in Jiangsu, 87% in Henan and 88% in Heilongjiang in order to breakeven in the first year. Assuming a 30% reduction in fertilizer drops the necessary yield increases by 4-8% in all cases apart from Yunnan, where fertilizer use is so low that even a 30% reduction has no impact on the break-even yield requirement. In fact the model predicts that biochar prices must drop by 2-8 times the current market value in order to become profitable (Table 3.8).This reduction is very large, even when acknowledging the early stage of biochar development and the possibility of future technology and biochar price reductions (Bridgwater, 2009).

Table 3.8: Agronomic value of biochar (USD kg^{-1}) at 10% and 20% yield increases and 10t ha^{-1} application rate

	Heilongjiang	Henan	Jiangsu	Yunnan
10%	0.04	0.03	0.05	0.06
20%	0.07	0.07	0.10	0.13

Overall these findings suggest that widespread adoption of commercial biochar as a standalone product requires significantly cheaper market prices than are currently available. The next section therefore examines the economics of on-farm biochar production using small- and medium-kiln technologies.

3.5.2 On-farm feedstock availability for making biochar

Biomass availability for on-farm biochar production was calculated using self-reported data on crop yields, published literature on product-residue ratios of grain:straw (Zhang and Zhu, 1990) and farmer-reported uses of agricultural residues in each case study. The analyses focus solely on agricultural straws, as other biomass sources were unavailable, i.e., maize cobs were used as fuel and rice husks were retained by the rice-husking mills.

Figure 3.3 displays the reported uses of straw in each case study. Where straw was fed to animals or used for energy (indicated by solid white or black bars), it was assumed unavailable for biochar production. Where straw was returned to the soil, given away, burned, or sold (indicated by hatched bars) it was assumed available for biochar production.

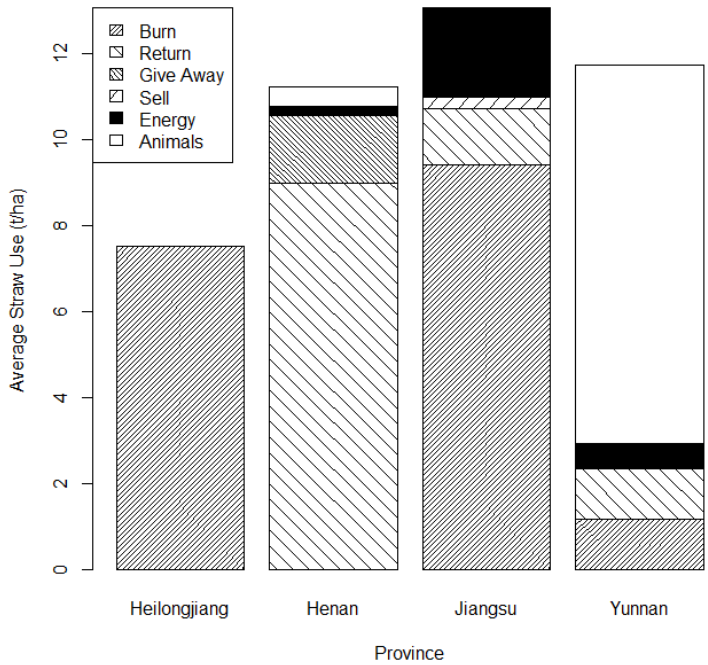


Figure 3.3: Cumulative bar chart of straw availability and use (t ha^{-1})

Yunnan has the lowest biomass availability (2.3 t ha^{-1}) due to the high numbers of draft and cash-generating animals that make up an essential part of its farming system. In contrast Heilongjiang, Henan, and Jiangsu have significant volumes of spare straw that could be used to make biochar. If all available straw was converted to biochar the equivalent annual biochar application rates would be 2.5, 3.5, 3.7, and 0.8 t ha^{-1} in Heilongjiang, Henan, Jiangsu and Yunnan, respectively.

3.5.3 On-farm production of biochar

The small kiln was tested within the Henan, Jiangsu and Yunnan models, however its high labour requirement meant that yield increases of over 100% were necessary to breakeven. These were deemed infeasible and therefore the remainder of this paper will consider the medium kiln scenario tested within Heilongjiang, Henan and Jiangsu. Figure 3.4 displays the break-even yields required at each site.

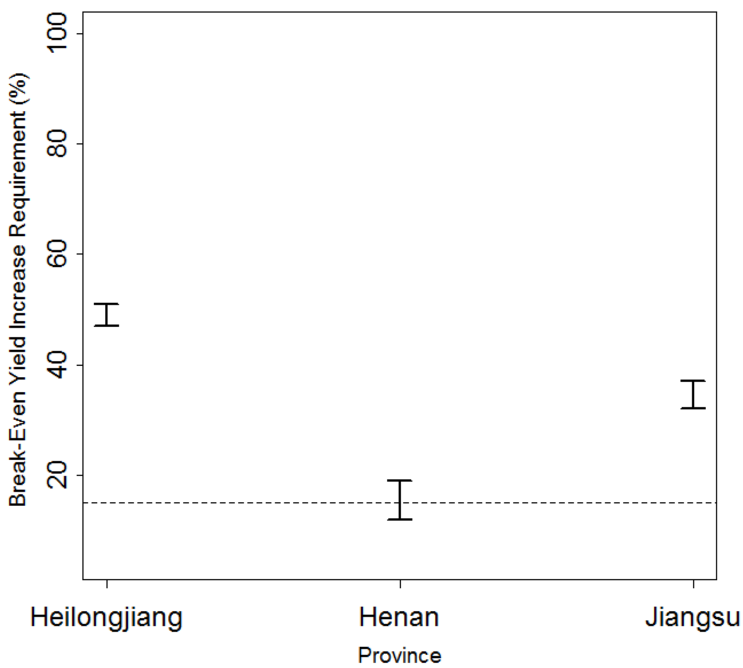


Figure 3.4: Medium kiln break-even yield increase requirements compared to average biochar effect

Heilongjiang has the highest break-even yields, due to a high cost of labour (Table 3) and its single cropping season. If we assume that one application of biochar has yield impacts over two years (without re-addition) the break-even yields for commercial biochar drop to 20-22% and for the medium kiln drop to 18-26%, however these are still on the high end of yield increases that might be expected.

Moreover there are significant practical difficulties in using a 500kg kiln on 25ha farms producing around 188 tonnes of straw annually, and requiring 375 kiln runs for complete pyrolysis of this straw to biochar.

Jiangsu's break-even yields are the next highest, and are greater than Henan despite comparable input costs, grain sale prices and labour rates. This is because straw in Jiangsu can be sold to local paper mills or rope factories for US$0.04kg^{-1}, creating a competing use that the agronomic value of biochar must exceed. Removing the option to sell straw reduces Jiangsu's breakeven range to 20-25%, however this is still high when considering the high soil fertility and ability of farmers to afford sufficient inorganic fertilizer. Moreover plentiful off-farm work in Jiangsu makes the relatively high risk strategy of biochar incorporation less appealing than the guaranteed wage-earning potential available in factories. Overall it seems unlikely that on-farm production of biochar would be adopted in sites like Jiangsu, where off-farm work is the main source of income and farming has low economic importance.

Henan's breakeven range is the lowest, and straddles the 15% yield increase point. Importantly, these results are generated assuming the same labour calculation as for other sites. However Henan is unique in that straw burning prohibition is strictly enforced in parts of this province, in contrast to Heilongjiang, Jiangsu and Yunnan. Therefore households must manually remove their straw from the fields after harvest, facing steep fines where the policy is disobeyed. Discounting the straw collection time (1.8mins kg^{-1}) from the labour estimate for the medium kiln (3.7mins kg^{-1} biochar created and spread) gives a Henan-specific labour time of 1.9mins kg^{-1} biochar created and spread. This reduces the breakeven range to just 1-2%, which suggests some potential for biochar as an agronomic input in

Henan.

Attention now turns to the time required to pay back the capital costs of the medium kiln, focusing on Henan as the likely candidate for adoption. The key consideration is whether sufficient profits can be generated over and above the capital cost of the medium kiln to justify investment. Profits are calculated based on the average land allocation in Henan (0.46ha) and assuming that farmers must take a loan in order to pay the kiln's capital cost (US$400). The Chinese base rate of interest at the time of publication is 7%, and is used to amortise payments over the 5-year estimated life-span of the kiln, creating an annual payment of US$97.1. Profits are calculated according to biomass available for biochar production (10.5 t ha^{-1} yr^{-1}) and the value of additional grains grown under a range of possible biochar yield increase values.

Three scenarios are investigated: a baseline (B), a best case scenario (BC) and a low labour (LL) scenario. In the B scenario, values of grain sales, labour hire and fertilizer are as they were at the time of the survey, and farmers are assumed to be able to use 10% less fertilizer per 10t ha^{-1} of biochar that they apply. In the BC scenario, grain sale prices increase 20%, fertilizer prices increase 20%, and farmers use 20% less fertilizer for a 10t ha^{-1} application. The LL scenario is the same as the B scenario, except that straw collection time is not included in the labour estimates, due to the enforced straw collection that was observed in the Henan case study. Figure 3.5 displays the 5-year net profit for these scenarios, over a range of hypothetical yield responses. Biochar is re-applied annually, with each applications effects lasting one year.

The BC and B scenarios require 20-30% and 30-40% yield increases respectively

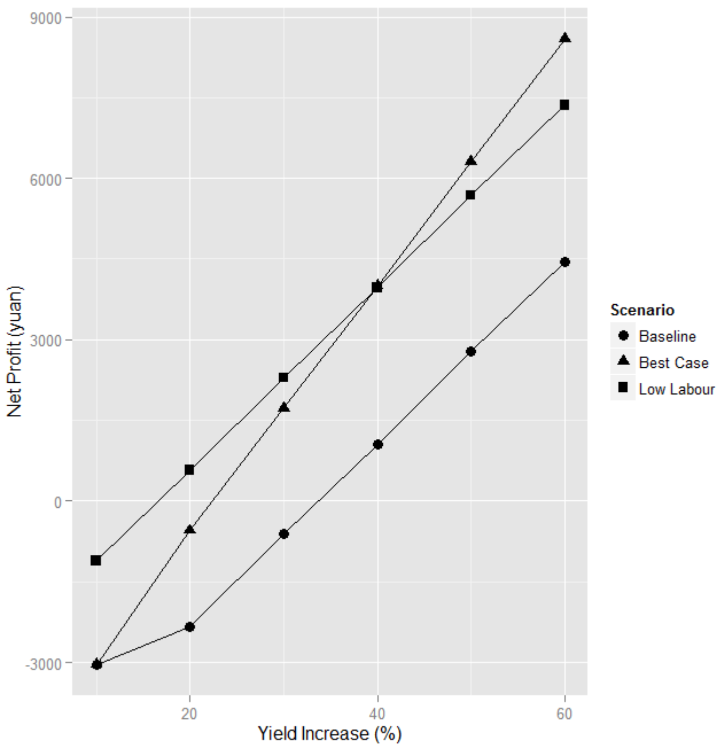


Figure 3.5: Net profit over five years on a 0.46ha Henan farm, at varying yield increase percentages, assuming biochar application every year

to create net profit, whereas the LL scenario is profitable at 18% yield increases. This is just above the average yield increases reported in published meta-analyses (Jeffery *et al.*, 2011; Biederman and Harpole, 2013; Liu *et al.*, 2014a) however the economic gains necessary to spark adoption are likely to require much higher yield increases. For example, assuming that annual biochar application creates and maintains a 20% or 30% yield increase, an average Henan household would gain just US\$18.2 year^{-1} or US\$73.1 year^{-1}, respectively, in net profit, after their annual loan payment. These figures equate to approximately 1.2 or 4.8 days of casual labour respectively, and are therefore relatively low sums of money, both in comparison to casual work, and also when considering the significant outlay of time and risk required to apply for a loan and purchase the medium kiln.

However all estimates displayed in Figure 3.5 assume that biochar's effects are only one year in duration, which is likely to be an underestimate of biochar's long-term impacts on yield. Therefore analyses were also conducted assuming that biochar's effects persist for two years on the basis of a single application (Figure 3.6). In this case, the two-year profit for a typical Henan farm was calculated, from which a one year average was derived, and then multiplied by five to provide an average five-year net profit figure.

Under these assumptions, the B scenario now requires 25% yield increases to break-even, and generates US\$147.2 net profit over five years at 30% yield increases. The LL scenario earns US\$108.2 net profit over five years at 20% yield increases. Once again, these scenarios do not provide a convincing economic case for farmers to invest in the medium kiln. In fact, only the BC scenario produces a substantial five-year net profit, and must assume a 20% increase in grain sale and fertilizer purchase price, maintenance of the same labour cost, and 20% reduction

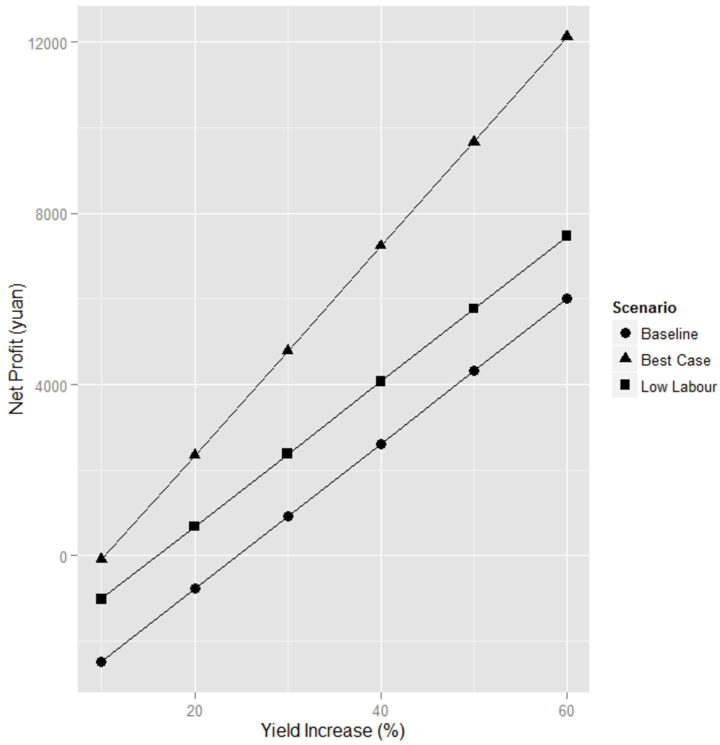


Figure 3.6: Net profit over five years on a 0.46ha Henan farm, at varying yield increase percentages, assuming biennial biochar application, with agronomic benefits maintained throughout

of fertilizer use. The likelihood of all these conditions coinciding seems unlikely, again casting doubt over the economic feasibility of the medium kiln in Henan.

3.6 Discussion

Despite much discussion on the potential of biochar as an agronomic win and/or poverty alleviation strategy for smallholder farmers in developing countries (Carter and Shackley, 2011; Pratt and Moran, 2010; Whitman *et al.*, 2011) we find that biochar's economic benefits are only likely to significantly exceed its costs in one of the four case studies examined here, and even then only under "best case" economic assumptions. This result is driven by a variety of factors. Firstly, the current price of commercially-produced biochar in China is too high to justify its inclusion as an additional input to inorganic fertilizer, and requires yield increases ranging from 43% (Yunnan) to 88% (Heilongjiang) in order to break-even in the first year after application. Whilst such yield increases do exist in the research literature (Kimetu *et al.*, 2008; Major *et al.*, 2010) they have not been reported in published Chinese field trials, and far exceed the average range reported by meta-analyses (Jeffery *et al.*, 2011; Biederman and Harpole, 2013; Liu *et al.*, 2014a). Thus we conclude that commercially-produced biochar is unlikely to succeed in China, when applied as a separate product in addition to inorganic fertilizer/other farm inputs. Exceptions to this may exist where biochar is used on crops with high market values, where soils are particularly degraded, or where biochar is part of an organic amendment product/system that commands a market premium.

For biochar produced on-farm, the production capacity of technologies and resultant labour demands per unit of biochar are very important. The small kiln is only capable of pyrolysing 10kg of straw per run, and the resultant high labour requirements render it unprofitable without a minimum doubling of yields for a 10t ha^{-1} application.

By contrast the medium kiln can pyrolyse 500kg of straw in one run, with significantly lower labour requirements. However even this relatively labour-efficient model is only profitable in one of the four case studies (Henan province) where break-even yields for labour cost alone are 12-19% if labour for straw collection is included, and 1-2% where it is excluded (justified by the mandatory collection of straw that is already enforced as part of a ban on straw burning). These results seem promising, but the kiln's high capital cost means it can only generate significant profits over five years where yield increases are consistently 20% higher than baseline on the basis of a biennial application of biochar; where grain sale prices and fertilizer prices increase by around 20% from baseline; and where fertiliser use is reduced by 20%.

Overall it seems unlikely that these optimal economic circumstances will combine to elevate the medium kiln to the level of profitability that is likely to be necessary, at a minimum, for the adoption of a new agricultural technology such as biochar. Moreover, even if the medium kiln was profitable at this static point in time, China's fast-moving development means that on-farm labour will become increasingly scarce and any on-farm economic gains are likely to be eclipsed in the face of off-farm employment opportunities. It is therefore essential for biochar technologies to adapt quickly in the face of such change. Specifically, researchers might turn their attention to the concept of biochar-mineral-chemical-composites (BMCCs).

3.6.1 Biochar-mineral-chemical-composites

One opportunity to reduce the labour requirements and adoption barriers to biochar use in China is to combine biochar with inorganic-fertilizers, creating BMCCs. In this scenario, straw collection is mechanised, transported to a central depot, pre-processed, pyrolysed to biochar and then combined with inorganic fertilizer. This system for collection and pyrolysis is already in place in at least two factories in Henan, however they currently sell biochar as a separate farming input, at a price that this study suggests is unlikely to be profitable for local farmers. However, biochar sold at this price could be a highly cost-effective component of a BMCC.

For example, recent research has begun investigating the potential of biochar-mineral-chemical-composites (BMCCs), which combine biochar with other substances in order to maximise its efficacy in soil (Joseph et al., 2013). In a Chinese context, where fertilizer overuse and open-field straw burning are prominent environmental issues, this presents an important opportunity. Specifically, the US\$0.26kg^{-1} biochar price is below that reported for compound fertilizers across our case study sites (US\$0.5-0.64kg^{-1}). This raises the possibility that, in a standard 40kg bag of inorganic fertilizer, perhaps 10kg of each bag could be replaced with biochar, creating a BMCC that is cheaper weight-for-weight than inorganic fertilizer. Where farmers are over-applying fertilizer, this reduction in fertilizer application rate is unlikely to harm crops (Zhang *et al.*, 2013c), whilst addition of biochar to the fertilizer compound may slow nutrient release, improve soil quality and increase soil organic carbon levels over time, thereby improving the sustainability of farming practices. Moreover, whilst attempts to encourage lower fertiliser use in China have been both expensive (requiring significant extension contact time with farmers) and had little long-term success (Huang *et al.*, 2008), application of a BMCC product would require no change in farmer practices: the same weight of product would be applied to the field, thus requiring simply a replacement of one product for another, rather than a change in behaviour. Finally, the development and sale of BMCCs through state-linked fertilizer companies could prompt rapid and broad-ranging geographical distribution across China, particularly if the government saw potential in biochar to meet their recently outlined carbon intensity reduction targets (Yuan and Zuo, 2011).

3.6.2 A shift in the biochar debate

From the discussion above, it seems unlikely that biochar will be an attractive agricultural technology for Chinese farmers under the current high-rate application approach. Moreover the characteristics of farms for which biochar may currently be profitable are likely to change as China's industrial development continues. Thus biochar researchers should consider studying BMCCs in order to produce products relevant to the vast majority of China's farmers. This will generate new questions about the optimal mix of inorganic fertilizer and biochar compounds, but most importantly it necessitates a re-framing of the idea of biochar as a yield increasing and/or poverty alleviating technology.

Yield increases
Biochar's relationship to yield increases has been the cornerstone of biochar research since it began. However, in the context of BMCC products, the emphasis could shift away from increasing yields to simply maintaining yields. Where the cost of a combined biochar-fertilizer product can undercut the price of pure inorganic fertilizer, the onus is on proving that replacing a portion of the fertilizer with biochar will at least maintain yields, rather than necessarily increase them. With small additions of biochar, yield effects arising from soil structure or SOM improvements could take time to appear, but this may not matter if farmers

are assured yield maintenance using a cheaper product. Moreover, this model of small, regular applications of biochar alongside other nutrient sources is arguably a much better analogue to our understanding of how *terra preta* soils were formed, rather than the high-rate application field trials that typify current biochar research.

Poverty reduction

The idea of biochar as a poverty-reducing agricultural technology has proved a popular concept, however evidence for its capacity in this regard is scant. In a Chinese context, it seems biochar has limited capacity to reduce household poverty. Firstly, small-scale technologies (such as the small kiln assessed in this chapter) that poorer households are able to afford require an estimated doubling or more of grain yields in order to outweigh the labour costs alone. Such increases are unlikely, even on highly degraded soils. Moreover reliance on agricultural residues as fodder for draught animals and as a household energy source means that poorer households have limited spare biomass available for biochar production. The combination of these constraints is likely to exclude poorer smallholders from on-farm biochar production unless support schemes can assist with capital costs of larger units, or unless community-level, large-scale pyrolysis units are organised. No published research exists on the feasibility of such avenues, and overall it seems pragmatic to focus instead on disseminating BMCC products that could reach these farmers through existing fertilizer distribution channels, without the need for increased labour and/or technology investment.

3.6.3 Conclusions

Overall, the application of biochar from commercial sources, small- and medium-kiln technologies is unlikely to increase income for farmers across a range of agricultural systems in China, suggesting that few farmers are likely to adopt biochar either for its agronomic gains or poverty alleviation potential. Overall this casts doubt on biochar's win-win potential, which hinges on its attractiveness to farmers as an agricultural input.

Although BMCCs are an intuitively appealing prospect for biochar in China, there is currently a lack of detailed research work and controlled field trials on their agronomic and soil impacts. In the absence of this information, it is not possible to pursue socio-economic evaluations of this concept further. However, there is scope for further investigation of biochar's potential in China from a commercial-production perspective. Despite concluding in this chapter that commercially-produced biochar is not an attractive agronomic input for farmers, the pricing strategy for this biochar and its relation to production costs were unclear during interviews with the pyrolysis company in Henan (Sanli New Energy). Therefore a more detailed investigation of the business case for biochar produced from China's straw residues is warranted, and this forms part of the research contained within the next chapter.

Acknowledgements

This work was supported by the Policy Innovation Systems for Clean Energy Security (PISCES) programme and the Sustainable Agriculture Innovation Network (SAIN), both funded by the UK Department for International Development (DFID). It was also funded by the EPSRC (Science and Innovation Award) and Scotland's Rural College (SRUC).

The authors would like to thank Professor Pan Genxing at the Nanjing Agricultural University, the Kunming Institute of Botany, the Chinese Academy of Agricultural Engineering and Meng Qingnan for their in-country support, and Stephen Joseph for his very helpful comments. The views expressed in this paper are entirely those of the authors.

Chapter 4

Competing uses for China's straw: business, government and climate mitigation perspectives

This chapter has been published online as:

Clare, A., Shackley, S., Joseph, S., Hammond, J., Pan, G., & Bloom, A. (2014). *Competing uses for China's straw: the economic and carbon abatement potential of biochar.* Global Change Biology Bioenergy, doi: 10.1111/gcbb.12220

Author Contributions

Abbie Clare: Formulated the concept for the paper, undertook interviews, collected technical and economic data from academic and technical publications, constructed the LCA and CBA models, performed Monte Carlo analyses, interpreted results and wrote the paper

Simon Shackley: Provided overall guidance on paper concepts, and gave suggestions for the final draft of the manuscript

Stephen Joseph: Provided suggestions regarding technical parameters of gasification and pyrolysis units

James Hammond: Participated in early discussions about the paper concept and provided feedback on a first draft of the paper

Genxing Pan: Facilitated contact for interviews with Sanli New Energy Company, and provided relevant background information on biomass management in China

Anthony Bloom: Assisted with questions regarding Monte Carlo analysis and coding in R

4.1 Chapter Rationale

The results from Chapter 3 suggest that biochar's agronomic benefits are unlikely to outweigh the costs of sourcing/production and application (thesis objective 1) for farmers across a range of China's agricultural systems. However, an absence of detailed data on the costing of commercial biochar in China, and the possibility that economies of scale may lead to a more financially viable biochar product than that made by farmers, begs the question of whether biochar production might be an attractive business prospect in China (thesis objective 2). Moreover, it is unclear what, if any, governmental/carbon pricing support a commercial biochar operation in China might need in order to be financially viable both in its own right and in comparison to other commercial uses of China's straw feedstocks, and further whether this support represents good "value-for-money" from a carbon mitigation cost-effectiveness perspective (thesis objective 3). Life-cycle analysis, social and financial cost-benefit analysis methods are used to answer these questions in this chapter.

4.2 Introduction

In the next two decades, China must increase gross agricultural productivity by an estimated 30-50% to keep pace with a growing population and their progressively resource intensive diets (Zhang *et al.*, 2011). Moreover, it must achieve this on arable land that is diminishing in size and fertility due to industrial-contamination of soils (Chen, 2007) and which suffers from low soil organic matter levels (Pan, 2008; Fan *et al.*, 2012). Additionally China needs to tackle the current wide-spread overuse of chemical fertilisers and pesticides, which is leading to significant eutrophication of water bodies (Zhang *et al.*, 2013a), alongside substantial air pollution and associated climate change from anthropogenic emissions of reactive nitrogen (Liu *et al.*, 2013).

In principal, biochar is a technology that may be able to address many of China's agricultural challenges, particularly as China appears to have soils upon which biochar's impact on crop yields may be most significant, as demonstrated in a recent global meta-analysis of biochar studies (Crane-Droesch *et al.*, 2013); research on the decline of SOC in China's soils, particularly on non-paddy land (Lal, 2002; Tang *et al.*, 2006); and many China-based agronomic trials (Zhang *et al.*, 2010b; Bian *et al.*, 2013; Lashari *et al.*, 2013).

Additionally, existing biochar systems analyses report strong economic and environmental preferences for the use of waste biomass materials as biochar feedstocks, rather than using wood or other virgin biomass (Roberts *et al.*, 2010; Shackley *et al.*, 2011b). China demonstrates significant potential in this regard, producing an annual 800 million tonnes of agricultural straw residues, of which an estimated 505 million tonnes are available after retaining sufficient straw to maintain soil quality (Jiang *et al.*, 2012). Moreover, many studies report that high proportions of straw are burned in-field. For example, Wu *et al.* (2001) report that 33% of crop straw was burned in Jiangsu province, compared to 32.4% for Guangdong province (Lin and Song, 2002), 40% for Fuzhou city (Yu, 2003), and 39.6% for Shanghai (Yao *et al.*, 2001). This is a consequence of low mechanisation rates (Tang *et al.*, 2006) and farmer demographic characteristics (Cao *et al.*, 2006) with farmers of greater income tending to burn more straw because of reduced demand for straw as a household fuel or for animal fodder, and a scarcity of on-farm labour for straw collection. This in-field straw burning emits high levels of particulate matter (PM), hydrocarbons and other pollutant gases to the atmosphere, resulting in significant local and regional air quality deterioration (Duan *et al.*, 2004; Yan *et al.*, 2006).

However, despite currently being plentiful, these straw residues are increasingly in demand as a result of China's bioelectricity subsidies. Recognising the adverse environmental and health consequences of in-field straw burning, the Chinese government is providing financial incentives to promote the mechanised collection and conversion of straw to electrical energy that is fed into the national grid. The financial incentives offered are structured as a feed-in-tariff (\$0.12 kWh^{-1} produced from agricultural and waste forestry biomass), subsidised loans, tax breaks and/or grants (Zhang *et al.*, 2014). The feed-in-tariff rate is comparable to western bioenergy policies, (for example, UK energy companies can typically sell renewably-generated electricity for between \$0.08-0.25 kWh^{-1}), however opinion is divided on whether these incentives are sufficient to create economically viable bioenergy projects (Lu and Zhang, 2010a; Zhang *et al.*, 2013b, 2014).

In addition, the extent to which these bioenergy subsidies might affect the economic viability of biochar projects is unknown. This therefore raises questions about how biochar, which could compete with bioenergy for straw feedstocks, might compare economically for businesses and environmentally as a contributor to GHG emission reductions in China.

We therefore investigate and contrast the economics and carbon abatement potential of using China's straw resources for biochar production via pyrolysis with two bioenergy technologies: straw briquetting and straw gasification. These scenarios are compared against two reference cases (straw reincorporation and in-field straw burning) and are analysed in terms of their relative profitability from a business perspective, and in terms of their environmental benefits from a GHG balance perspective.

4.3 Materials and Methods

Cost-benefit analysis (CBA) is used to compare the economic viability (net present value (NPV) per oven dry tonne (odt) straw), and life cycle analysis (LCA) is used to compare the environmental ($MgCO_2e$ per odt straw) outcomes associated with three straw utilisation scenarios: straw briquetting and subsequent combustion for heat energy (S_{Briq}); straw gasification for electrical energy (S_{Gas}); and straw pyrolysis for biochar and electrical energy (S_{Pyr}). These are compared to two baselines of straw reincorporation (S_{Rein}) and straw burning (S_{Burn}). S_{Rein} assumes that all straw is incorporated into the field whereas S_{Burn} assumes that straw is burned in-field.

4.3.1 Technology Scenario Selection

Straw briquetting (S_{Briq}) was chosen as a comparison scenario based on observations of straw briquettes on sale in Chinese town markets and online. Briquetting has much lower capital and technological expertise requirements than gasification and pyrolysis, and is therefore likely to be perceived as lower risk by investors and as an accessible option for small businesses. However it does not qualify for government bioelectricity subsidies, as briquettes tend to be bought for local heat and cooking applications rather than burned for commercial electricity generation. In contrast, straw gasification (S_{Gas}) was chosen on the basis that gasification is identified as a priority bioenergy technology in Chinese national policy documents (Han *et al.*, 2008; Zhang *et al.*, 2014), has been implemented in many technological development projects across China (Kirkels and Verbong, 2011), and is reportedly a viable economic proposition for Chinese businesses (Lu and Zhang, 2010a). Although co-firing biomass with coal has also been found to be an economic use of straw residues (Lu and Zhang, 2010a), it was not included as an option because the Chinese government does not currently provide financial incentives for bioelectricity produced through co-firing. This is due to concerns over the accurate verification of biomass co-firing rates at existing coal-fired power stations (Gan and Yu, 2008; Dong, 2012).

The pyrolysis (S_{Pyr}) scenario investigates the use of slow pyrolysis technology to produce biochar and a relatively small amount of electricity. Slow pyrolysis delivers less electricity than other bioenergy options, because a proportion of the feedstock is converted to biochar and not into heat or electrical energy (Brown, 2009).

Each of the S_{Briq}, S_{Gas} and S_{Pyr} technology scenarios is guided by interviews conducted in summer 2012 at the Sanli New Energy bioenergy-plant in Henan Province, China. Sanli New Energy has capitalised upon the combination of a local straw-burning ban, related straw-burning avoidance subsidies (\$28 Mg^{-1} straw paid to businesses that use straw for livestock rearing, paper production or bioenergy generation) and national bioelectricity subsidies (Zhang *et al.*, 2014), to build a 4MW pyrolysis unit and straw briquetting plant. Data on Sanli's economics, straw collection system and size guided the choice of parameters used to structure and assess the S_{Briq}, S_{Gas} and S_{Pyr} scenarios. Table 4.1 provides an

Table 4.1: Overview of technical parameters for briquetting, gasification and pyrolysis

	Briquetting	Gasification	Pyrolysis
Technology Readiness Level (TRL)	9	7-8	5-6
Operation Lifetime (yrs)	20	20	20
Straw processed (odt yr^{-1})	28,000	28,000	28,000
Annual output	28,000 Mg briquettes	26,680 MWh bioelectricity	8,400 MWh bioelectricity; 8,300 Mg biochar
Energy offset	Equivalent MJ heat energy from coal briquettes	Equivalent MWh electrical energy	Equivalent MWh electrical energy
National bioelectricity subsidies	None	Feed-in-tariff; subsidised loans; tax breaks;	Feed-in-tariff; subsidised loans; tax breaks;
Local straw-burning subsidies	Avoided straw burning	Avoided straw burning	Avoided straw burning

overview of these parameters. More detailed information on technology configuration is available in the Appendix 2 (S9-S17, and Figures S1 and S2).

The Technology Readiness Levels (TRL) for each technology (straw briquetting, gasification and pyrolysis) are also estimated, based on expert opinion and observations of the deployment of these technologies in rural Chinese settings. A TRL is a scale from one to nine that indicates the maturity of a given technology (Mankins, 1995; UK Ministry of Defence, 2014). Table S1 in Appendix 2 provides a description for each TRL. Briquetting scores the highest (9), as a mature off the peg technology, followed by gasification at stages 7-8, and then pyrolysis at stages 5-6.

4.3.2 Cost Benefit Analysis

Published literature, industry reports, policy documents, interviews and online market estimates were used to develop appropriate pricing structures for S_{Briq}, S_{Gas} and S_{Pyr}, adjusted to 2014 prices. The CBA combines these values to generate an estimate of scenario profitability from the perspective of a business or potential investor, taking account of government bioelectricity and avoided straw burning subsidies.

The agronomic value for biochar is estimated by combining data on the microeconomics of farms in Henan (presented in Chapter 3) with data from the latest published meta-analyses on biochar's yield impacts (Jeffery *et al.*, 2011; Crane-Droesch *et al.*, 2013), the findings from which are also consistent with results from China-based biochar experiments (Wang *et al.*, 2012a; Zhang *et al.*, 2012a). Biochar's agronomic value is calculated as the value of the yield improvement seen in one growing year, per unit of biochar applied, assuming that biochar is applied once and that its effects last across two growing seasons. It should be noted that this estimate does not take spreading and transportation costs into account, and that therefore the commercial sale price of biochar to farmers will need to be less than this figure. The baseline agronomic value for biochar of \$110Mg^{-1} is calculated according to the latest meta-analysis by Crane-Droesch *et al.* (2013), who report a 10% yield increase for a 3Mg ha^{-1} application rate. However, the more conservative estimate of Jeffery *et al.* (2011), assuming that a 10Mg ha^{-1} application stimulates 10% yield increases, gives biochar an agronomic value of just \$33Mg^{-1}. This is a significant price difference, and therefore the retail price of biochar is varied in the sensitivity analysis, reflecting this uncertainty and investigating the extent to which it impacts the overall profitability of S_{Pyr}.

The briquette market value is calculated based on the typical energy density of straw briquettes (McKendry, 2002a; Roberts *et al.*, 2010) and the value of this energy is based on the spot price of coal in China at the time of writing (\$95 Mg^{-1}) (Zhao and Yan, 2012; Bloomberg, 2014). Finally, the market value of bioelectricity is set in line with the current Chinese bioelectricity subsidy of \$0.12 kWh^{-1} (Zhang *et al.*, 2014).

The NPV of each scenario is calculated at the project level, over a 20 year lifetime, taking subsidised loans and tax breaks into account where relevant (Zhang *et al.*, 2014). The discount rate is set at 3.5% (Federal Reserve Bank of St Louis, 2014).

4.3.3 Life-Cycle Analysis

A GHG-oriented attributional LCA was performed, based on the ISO 14040 (ISO, 2006) guidelines, and using a 100-year global warming potential. The three main GHGs were accounted for (carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)), and these are henceforth displayed in terms of their carbon dioxide equivalent global warming potential (CO_2e), calculated according to IPCC guidelines of CO_2e equivalence as 25 for CH_4 and 298 for N_2O (IPCC, 2007). The GHG abatement potentials of S_{Briq}, S_{Gas} and S_{Pyr} were calculated using S_{Rein} as the baseline scenario, however the S_{Burn} scenario is also displayed for reference.

The analysis initially focuses on directly-attributable CO_2e emissions from each phase of the life cycle (raw material acquisition, production, distribution, energy offset and dismantling processes) before moving on to consider the indirect CO_2e abatement potential of reduced soil N_2O emissions and avoided fertiliser use as a result of biochar application.

Soil N_2O reductions following biochar application have been widely debated for some years, however a recent meta-analysis (Cayuela *et al.*, 2013) provides greater clarity on the extent of this effect. Cayuela *et al.* (2013) report that biochars derived from woody and herbaceous feedstocks, including agricultural straws, demonstrate the highest emission reduction potential, with a 27% reduction in N_2O emissions for a 1-2% (by soil weight) biochar application rate. Data from this study is then combined with a China-specific field trial demonstrating a similar effect (Zhang *et al.*, 2012a) in order to calculate the additional contribution that N_2O emission reduction may have on the S_{Pyr} LCA result.

A similar approach is taken to calculating additional GHG abatement as a result of avoided fertiliser application. Recent trials in China suggest that the application of a combined biochar-NPK-clay compound (a biochar-mineral-chemical-composite (BMCC)) may be an economic option for farmers, where 25% of NPK is replaced by biochar, on a weight basis (Joseph *et al.*, 2013). This data is combined with data on the carbon intensity of China's domestic fertiliser production industry, which emits 13.5MgCO_2e MgN^{-1} fertiliser as compared to an average of 9.7 MgCO_2e MgN^{-1} in Europe (Zhang *et al.*, 2013c). The nitrogen (N) fertiliser is assumed to contribute to a standard NPK (16:16:16) mix. Emissions from potassium (K) and phosphorus (P) production in synthetic fertilisers are excluded, as they are an order of magnitude lower (West and Marland, 2002). Figure 4.1 displays the processes included in the direct and indirect abatement potential calculations.

The CO_2e offsets from avoided fossil fuel energy are calculated according to the carbon emission factor (CEF) of the fuel that straw-derived bioenergy is expected to replace. Straw briquettes are assumed to replace coal briquettes that are typically burned for heat and/or cooking purposes in local applications such as homes, schools and hospitals. In S_{Gas} and S_{Pyr}, each MWh of bioelectricity produced is assumed to replace one MWh of electricity in the central grid, which services Henan province and has an estimated carbon intensity of 1.13MgCO_2e MWh^{-1} (World Resources Institute, 2014).

Details of the GHG emissions associated with different phases of the lifecycle are given in Appendix 2 (S9-S17). Many of the parameters used to estimate these emissions are considered uncertain, therefore published literature and expert opinion were also used to estimate the uncertainty range and probability distribution of each parameter. An uncertainty analysis was then undertaken

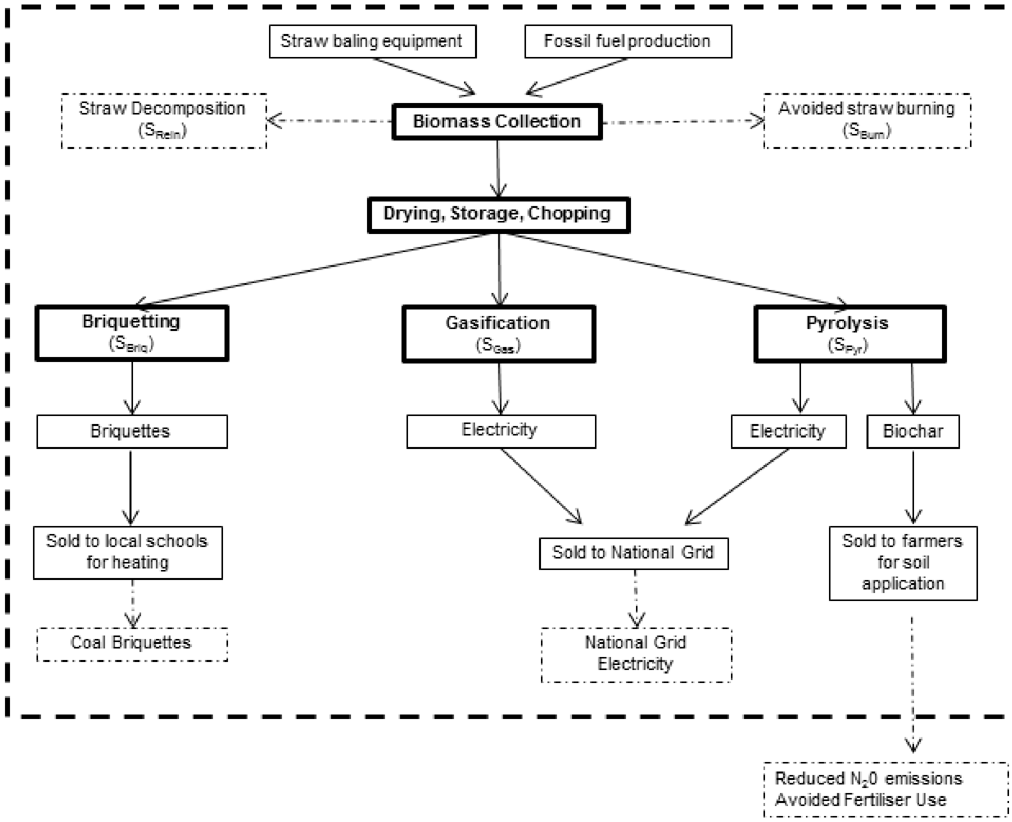


Figure 4.1: Diagram of LCA boundaries: bold boxes indicate processes that emit CO_2e, dashed boxes indicate CO_2e offset or abatement processes. Processes within the bold dashed line are considered direct impacts of each scenario, and processes outside the bold dashed line are considered indirect impacts

using a Monte Carlo method. 10,000 simulations were performed to derive median points and 95% confidence intervals for $MgCO_2e$ emitted per odt feedstock. The impact of each parameter's value on the final result was investigated using sensitivity analysis.

4.4 Results

4.4.1 Economic viability of briquetting, gasification and pyrolysis

Removing both national bioelectricity and local avoided straw-burning subsidies renders S_{Briq}, S_{Gas} and S_{Pyr} unprofitable, with project NPVs of \$-2.88 million (m), \$-19.0m, and \$-20.3m, respectively (see black bars in Figure 4.2). When including local avoided straw burning subsidies (see grey bars in Figure 4.2), S_{Briq} becomes profitable (NPV \$7.34m), whereas S_{Gas} and S_{Pyr} still generate significant losses (NPV \$-8.14m and \$-9.36m, respectively). However, the inclusion of income from China's national bioelectricity subsidy program (see white bars in Figure 4.2) has a significant impact on S_{Gas} profitability (NPV \$12.60m), increasing it above the unchanged S_{Briq} NPV (NPV \$7.34m). Meanwhile, S_{Pyr} remains unprofitable (NPV \$-1.84m), due to the relatively lower electricity volume yielded per odt straw by pyrolysis as compared with gasification.

However the NPV of S_{Pyr} is strongly influenced by the agronomic value of biochar, which is one of the most uncertain parameters modelled in this CBA. At the baseline agronomic value of \$110Mg^{-1}, (based on the results of Crane-Droesch *et al.* (2013)) the S_{Pyr} NPV (including all available subsidies) is \$-1.84m. However, assuming the more conservative agronomic value estimate of \$33Mg^{-1}, (based on

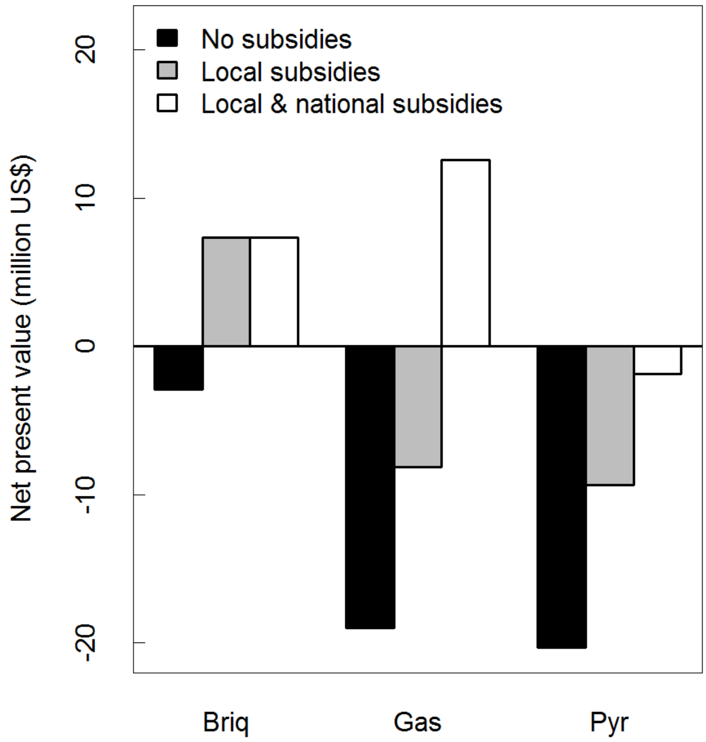


Figure 4.2: Net present value (million US$), with and without Chinese government subsidies, for S_{Briq}, S_{Gas} and S_{Pyr}

the results of Jeffery *et al.* (2011)) the S_{Pyr} NPV drops even further to \$-10.1m. For S_{Pyr} to break-even, biochar must sell for \$128Mg^{-1} if all other factors remain equal, or for \$206Mg^{-1}, if bioenergy subsidies are excluded. For the NPV of S_{Pyr} to equal that of S_{Gas}, biochar must sell for \$238 Mg^{-1}. Interestingly, in 2014 Sanli New Energy Company reported their biochar retail price as \$259 Mg^{-1}, which exceeds the break-even prices that are necessary for pyrolysis profitability. However this high sale price is at odds with the current understanding of biochar's agronomic value in soil (as outlined above) and studies on agricultural economics and farmer-perspectives of biochar in the area (Clare *et al.*, 2014).

4.4.2 Direct CO_2e abatement potential of briquetting, gasification and pyrolysis

Figure 4.3 outlines the CO_2e abatement potential of S_{Burn}, S_{Briq}, S_{Gas} and S_{Pyr}, including only direct processes in the analysis, all implicitly compared against S_{Rein} as the baseline scenario. The results suggest that, when including offsets from avoided fossil-fuel energy emissions (see black bars in Figure 4.3), S_{Briq} offers the greatest carbon abatement (1.35MgCO_2e odt^{-1} straw) followed by S_{Gas} (1.16MgCO_2e odt^{-1} straw) and S_{Pyr} (1.06 MgCO_2e odt^{-1} straw). This carbon abatement potential increases by 0.04MgCO_2e odt^{-1} straw for each scenario, if referenced to the S_{Burn} baseline rather than S_{Rein}. Interestingly this means that, despite only receiving local and not national subsidies, S_{Briq} appears to offer the greatest CO_2e abatement potential. However S_{Briq} also displays the most variance in its carbon abatement, as a result of the wide variability in data available for comparing emissions from straw and coal briquettes in small stoves (Zhang *et al.*, 2000; Wang *et al.*, 2013b).

If emissions offsets from avoided fossil fuel use are not included (see grey bars in

Figure 4.3), both S_{Gas} and S_{Pyr} still provide some carbon abatement. In the case of S_{Gas} this is because approximately 20% of feedstock carbon is initially stabilised in the ashy char produced during the gasification process (Lu and Zhang, 2010b) with 90% remaining stable over the 100 year time-scale of this analysis (Cross and Sohi, 2013). In the case of S_{Pyr}, 50% of feedstock carbon is initially stabilised in the biochar, with 80% of that amount (39% of the initial feedstock carbon) still remaining in the soil after 100 years (Singh *et al.*, 2012; Crombie *et al.*, 2013). This persistence is a pertinent point, as it can be argued that offset fossil fuel emissions are not avoided for long, because the fossil fuel still remains to be consumed. From these perspectives, it can therefore be argued that S_{Pyr} offers a more permanent GHG reduction than the other options.

4.4.3 Indirect CO_2e abatement potential of pyrolysis

The application of biochar to agricultural land may contribute to the abatement potential of S_{Pyr} via indirect processes, which generally have a higher level of uncertainty and variability than the direct factors already discussed. This can result from reduced certainty regarding biochar's impact on a given outcome (i.e., in the case of biochar's effect on N_2O emissions) and/or because the process relies on human behaviour change (i.e., the reduction in fertiliser application, or the application of biochar to land). Indirect environmental consequences of biochar application have been variously reported in past LCA studies (Roberts *et al.*, 2010; Hammond *et al.*, 2011; Sparrevik *et al.*, 2013), but recent evidence has improved the evidence base for the effect magnitude that might be expected for a given biochar application rate. Specifically, two indirect effects that have received increased attention are reduced N_2O emissions from soil, and improved fertiliser use efficiency.

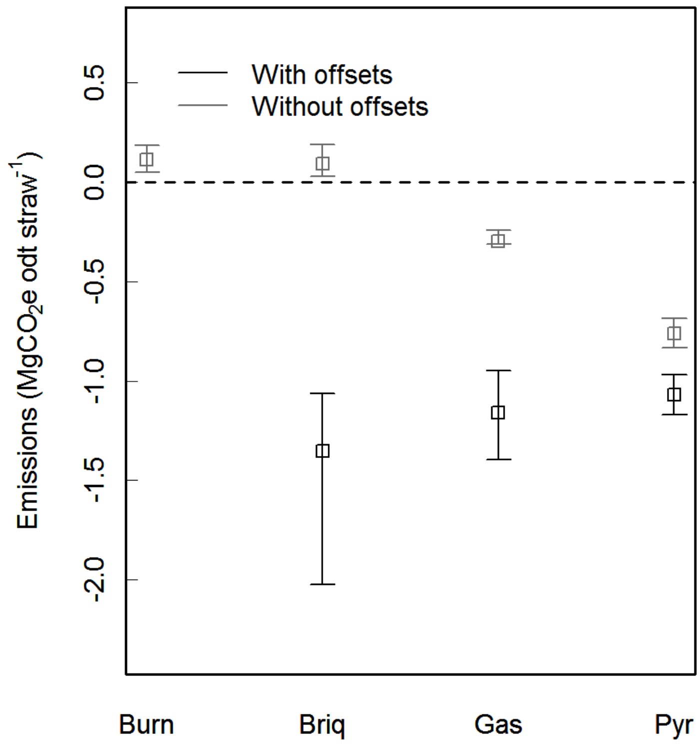


Figure 4.3: Median and confidence interval estimates of $MgCO_2e$ abated per odt straw processed in S_{Briq}, S_{Gas} and S_{Pyr}, including and excluding offsets from avoided fossil-fuel energy (black bars and grey bars, respectively). Uses S_{Rein} as the baseline, and displays S_{Burn} for reference.

4.4.4 Reduced N_2O emissions from soil

Table 4.2 combines data from a recent meta-analysis of biochar's impact on soil N_2O emissions (Cayuela *et al.*, 2013) with the baseline and reduced N_2O emission reductions reported in a China-based biochar field trial (Zhang *et al.*, 2012a). According to these data, and assuming a one-year effect of biochar on N_2O emissions, the abatement potential of S_{Pyr} could be increased by 0.004-0.012 $MgCO_2e\ yr^{-1}$. This represents a 1% increase in S_{Pyr}'s abatement potential, and we therefore suggest that the absolute contribution of biochar-induced soil N_2O emission reductions are relatively small.

Table 4.2: Calculations of avoided N_2O emissions per tonne feedstock pyrolysed

Biochar application rate (%)	0.5[1]	2[2]	1-2 [3]
% N_2O reduction from baseline	-40	-51	-27
N_2O avoided (kg odt^{-1})	0.021	0.007	0.007
Abatement potential ($MgCO_2e\ odt^{-1}$)	0.012	0.004	0.004

4.4.5 Improved fertilizer use efficiency

If biochar were to aid the reduction of fertiliser application in China, the resulting GHG mitigation potential is large. Using data from Joseph *et al.* (2013) and Zhang *et al.* (2013c) we calculate that each Mg of biochar that replaces chemical fertiliser could abate an additional 1.33$MgCO_2e$, and thus that each odt of straw feedstock being used to produce biochar could abate an additional 0.39$MgCO_2e$.

[1] Data taken from Zhang et al., (2012)
[2] Data taken from Zhang et al., (2012)
[3] Data taken from Cayeula et al., (2013)

Including these indirect effects of biochar application on avoided emissions from soil N_2O and fertiliser use reduction, the total abatement potential of S_{Pyr} increases to 1.46MgCO_2e odt^{-1} straw, which puts it ahead of both S_{Gas} (1.16MgCO_2e odt^{-1} straw) and S_{Briq} (1.35MgCO_2e odt^{-1} straw) in terms of carbon abatement.

4.4.6 Sensitivity analysis

Figures 4.4 and 4.5 graphically display the results of sensitivity analyses undertaken on key parameters influencing the NPV and carbon abatement potential, respectively, of the S_{Briq}, S_{Gas} and S_{Pyr} scenarios. Both figures present the baseline NPV/carbon abatement value and a surrounding range, calculated by varying key economic/carbon abatement parameters by +/- 20%, whilst keeping all other parameter values constant. The parameter values used in these sensitivity analyses are available in S19 and S20 of Appendix 2.

Figure 4.4 displays the influence of the following economic parameters on the overall NPV for each scenario: straw price, local straw burning subsidies, capital cost, labour cost, and the sale price of outputs (briquettes; electricity; electricity and biochar, for S_{Briq}, S_{Gas} and S_{Pyr} respectively). All NPVs displayed include the financial support currently available from both local and national subsidy programs.

The results in Figure 4.4 suggest that sales prices for output products are very influential on the overall economic viability of briquetting, gasification and pyrolysis projects. Likewise, varying the capital cost of pyrolysis and gasification units has a significant impact on the economic viability of S_{Gas} and S_{Pyr}, even

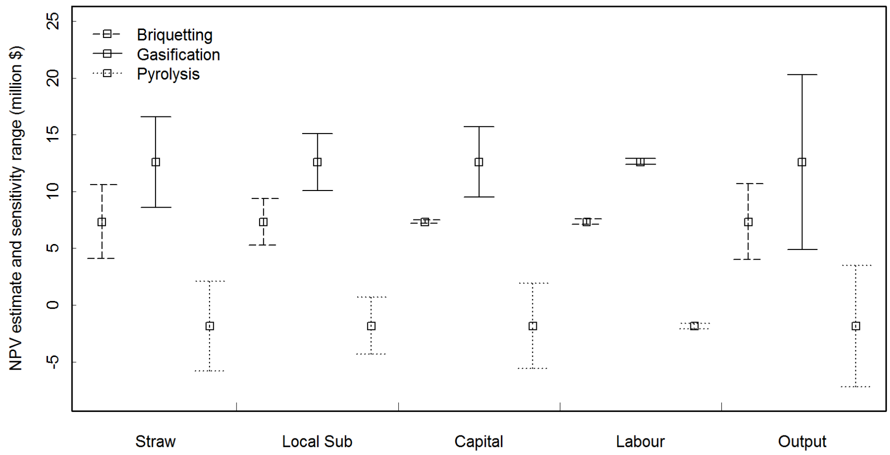


Figure 4.4: Baseline NPV estimates (million US$) and sensitivity analyses for key parameters determining the economic viability of briquetting, gasification and pyrolysis. Ranges are produced by independently varying key parameters (x-axis) by +/- 20% and recording the impact on the overall NPV value

tipping S_{Pyr} into profitability where capital costs alone decrease by 20%. This is particularly relevant when considering the early stage of technological readiness of pyrolysis and the subsequent drop in capital cost that might be expected as this technology reaches higher stages of maturity (Utterback, 1996; Shackley *et al.*, 2015). However it must also be noted that the top range of S_{Pyr}'s NPVs do not overlap with the bottom range of the NPVs of S_{Briq} or S_{Gas}, suggesting that pyrolysis will require significant improvements in multiple economic parameters before it can compete with briquetting or gasification.

Figure 4.5 displays the results of a +/-20% sensitivity analysis conducted on the following key parameters influencing the carbon abatement potential of S_{Briq}, S_{Gas} and S_{Pyr}: straw collection emissions; embedded emissions within machinery; direct emissions from the combustion of straw briquettes/ gasification of straw /

pyrolysis of straw; offset emissions from avoided fossil fuel energy; the stability of carbon sequestered within biochar; and offset emissions from avoided fertiliser use. The results suggest that direct emissions from combustion of straw briquettes/gasification of straw/pyrolysis of straw, and offset emissions from avoided fossil fuel use, have the greatest impact on the carbon abatement potential of each scenario.

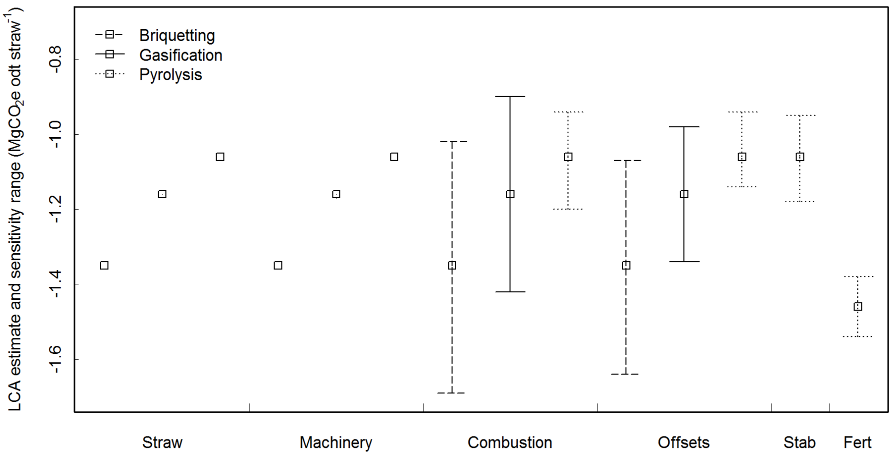


Figure 4.5: Baseline carbon abatement estimates (MgCO$_2$e abated odt straw^{-1}) and sensitivity analyses for key parameters determining the carbon abatement potential of briquetting, gasification and pyrolysis. Ranges are produced by independently varying key parameters (x-axis) by +/- 20% and recording the impact on overall carbon abatement potential

This suggests that gasification and pyrolysis units must be well designed, maintained and managed by staff with appropriate expertise, and that improvements to the efficiency of boilers that combust straw briquettes could also improve their carbon abatement potential. Variation in emissions from straw collection and machinery/building construction has a negligible impact on overall carbon abatement balance. However, variability in fertiliser use and the stability of carbon sequestered within biochar have modest effects on the overall carbon abatement potential of S_{Pyr}.

4.4.7 Carbon abatement cost-effectiveness

In light of the Chinese government's carbon intensity reduction targets, it is important to consider the cost-effectiveness of S_{Briq}, S_{Gas}, and S_{Pyr} in terms of CO_2e abatement. Our results show that all three technologies require assistance from carbon pricing in order to break-even, although S_{Briq} requires a significantly lower price than S_{Gas} and S_{Pyr} (see Table 4.3, where S_{Pyr}(D) includes only direct effects and S_{Pyr}(I) includes both direct and indirect processes discussed in this paper). Requiring a carbon price of \$7 $MgCO_2$e abated^{-1}, S_{Briq} is the only technology studied here that can produce carbon abatement for less than \$25 $MgCO_2e^{-1}$, as outlined in the Stern (2006) report. Moreover, early price indications from China's nascent emissions trading scheme (which currently covers five municipal areas and two provinces; (Lo, 2012)) suggest that domestic carbon prices (currently ranging between \$5 - \$20 $MgCO_2$ abated^{-1}) would only provide sufficient support to make S_{Briq} profitable (Song and Lei, 2014).

Table 4.3: Comparing CO_2e abatement cost effectiveness for briquetting, gasification and pyrolysis

	S_{Briq}	S_{Gas}	S_{Pyr} (D)	S_{Pyr} (I)
Subsidy required (\$ tonne feedstock processed^{-1})	5	34	36	36
Subsidy required (\$ $MgCO_2e^{-1}$ abated)	7	61	71	51

4.5 Discussion

Overall biochar struggles as both an absolute and relative win-win solution compared to alternative uses of China's straw resources. In the absolute sense, biochar remains an unprofitable business option under all the subsidy scenarios explored here, suggesting that it will take a significant level of subsidy support before businesses would consider producing biochar. In the relative sense, biochar is both less financially appealing and produces less climate change mitigation than the primary competing uses of straw biomass.

By contrast, straw briquetting for sale as a local fuel in heating and cooking appliances appears to be a win-win solution, both from the perspective of businesses and climate change mitigation. S_{Briq} has the greatest carbon abatement potential (1.35$MgCO_2e$ odt^{-1} straw as compared to 1.16 and 1.06$MgCO_2e$ odt^{-1} for S_{Gas} and S_{Pyr}, respectively), and the highest economic abatement efficiency (requiring a relatively small carbon price of \$7 $MgCO_2e^{-1}$ abated, compared to \$61 $MgCO_2e^{-1}$ or \$51-71 $MgCO_2e^{-1}$ abated, for S_{Gas} and S_{Pyr}, respectively.) Straw briquetting, as a mature technology, also has the highest technology readiness level (TRL), making it a relatively low risk proposition and subsequently attractive for small businesses and village level industry. This technology also leads to the direct use of biomass energy for heat in boilers and heating systems of local communities, thus negating the need for expensive equipment and avoiding the inevitable energy wastage when converting heat energy into electricity.

However the apparent success of straw briquetting is subject to two important caveats. Firstly, this scenario relies on the sale of straw briquettes to local households, schools and hospitals for combustion in relatively inefficient, small-scale boilers and stoves. However as China's energy system modernises, there may be a move towards more efficient district heating and power systems, which will reduce market demand for straw briquettes to be processed and sold in this way. Secondly, the heat energy produced from locally sold briquettes is not as fungible as electricity, which is socially a more highly valued commodity.

This is also reflected in the structure of the current Chinese bioenergy subsidies, which support bioelectricity generation through a feed-in-tariff, but provide no equivalent support for heat energy from the same biomass source. Consequently, this analysis finds that national bioelectricity subsidies increase the NPV of gasification (NPV \$12.60m) above that of briquetting (NPV \$7.34m), suggesting that the subsidies are set at a level that increases the profitability of electricity generation over that of heat generation from biomass. In contrast, pyrolysis remains unprofitable even when receiving local and national subsidy support (NPV \$-1.84m). For pyrolysis and associated biochar production to be able to compete with alternative uses of feedstocks, such as briquetting and gasification, the agronomic value of biochar will need to increase considerably. The current evidence suggests that biochar has an agronomic value of approximately \$110 Mg^{-1} in central, grain-growing Chinese provinces such as Henan (Crane-Droesch *et al.*, 2013; Clare *et al.*, 2014). However, these results suggest that biochar must sell for at least \$238 Mg^{-1}, in the presence of subsidies, for the NPV of S_{Pyr} to equal that of S_{Gas}. Moreover, the LCA analysis suggests that pyrolysis is unlikely to attract financial support from the Chinese government or climate change mitigation funds on carbon abatement grounds alone, unless the abatement potential of indirect processes such as avoided fertiliser use are included and can be further increased.

There are three considerations that may affect these findings. Firstly, regarding the indirect mitigation potential of avoided fertiliser use. In fact, fertiliser application rates in China are so high that fertiliser application can be reduced by up to

27% with no impact on yields, and without requiring biochar application (Huang *et al.*, 2008). This calls into question the necessity of biochar to stimulate this particular indirect carbon abatement mechanism because, although replacement of NPK with biochar to produce a biochar-mineral-chemical-composite (BMCC) could theoretically reduce fertiliser application rates (Joseph *et al.*, 2013; Clare *et al.*, 2014), biochar is not essential to achieving this goal.

Secondly, there are anecdotal reports of two factories in central China producing 60,000Mg year^{-1} of BMCC products for local agricultural markets (personal communication with Dr. Stephen Joseph (2014)). Field trials in China have recently suggested that BMCCs (which pre-mix low application rates of biochar with inorganic fertiliser and clay) can produce yield increases of up to 40% (Joseph *et al.*, 2013). Applying this data to agricultural market conditions in Henan province, biochar's value as a soil amendment would be \$5,740Mg^{-1}, increasing the S_{Pyr} NPV to over 50 times that of S_{Gas}. If these results are reproducible, this is a significant game-changer for the field of biochar research and application, however extensive field trials are necessary to ensure that such impacts can be replicated consistently.

Thirdly, the technological advancement, appropriate management and successful deployment of pyrolysis and gasification technologies will have an important impact both on their carbon abatement and economic potential. Improved technological maturity and deployment should improve the conversion efficiency from straw energy to electrical energy and/or biochar. This is a significant determinant of the overall economic viability and emissions balance of S_{Gas} and S_{Pyr}, both by increasing the units of economic output produced per unit of feedstock, and by avoiding emissions of strong climate forcing GHGs resulting from incomplete combustion. Also, the technological readiness of pyrolysis currently lags behind gasification, making it potentially more risky and less attractive for investors. As such, innovative technological developments are needed for pyrolysis technology to advance to a win-win position on economic viability and carbon abatement potential when compared to gasification and briquetting.

4.5.1 Conclusions

Overall, the findings of Chapters 3 and 4 call the environmental and economic win-win potential of biochar into question. It seems that biochar will struggle to be a profitable agronomic input for farmers (thesis objective 1), that it does not offer a financially attractive option for businesses (thesis objective 2), and that it has lower climate change mitigation potential and cost-effectiveness than competing uses of straw feedstocks (thesis objective 3). Chapter 4 also demonstrates that biochar struggles to compete with the economics of bioenergy, particularly under China's current feed-in-tariff for bioelectricity generation from agricultural residues.

In light of the economic and environmental benefits of using China's straw resources as a feedstock for bioenergy, an additional thesis objective was added to investigate China's bioenergy policy landscape. Specifically, it was noted that China's bioenergy feed-in-tariff is only available to bioelectricity units taking 80% or more of their feedstock energy from agricultural residues, effectively ruling out low energy replacement cofiring of agricultural residues with coal. Thus, thesis objective 4 was developed to explore how much bioenergy (TWh of electricity) can profitably be produced if China were to extend its feed-in-tariff for bioenergy to include low energy replacement ratio cofiring of agricultural residues in existing coal-fired power stations.

Acknowledgements

This work was supported by the Policy Innovation Systems for Clean Energy Security (PISCES) programme and the Sustainable Agriculture Innovation Network (SAIN), both of which are funded by the UK Department for International Development (DFID). It was also funded by the EPSRC (Science and Innovation Award) and Scotland's Rural College (SRUC). The authors would also like to thank Meng Qingnan for her assistance with data collection; Saran Sohi, Andrew Cross, and colleagues at the UK Biochar Research Centre for their constructive feedback; and the very helpful comments from anonymous reviewers of this paper.

Chapter 5

Should China subsidise cofiring to meet its 2020 bioenergy target? A spatio-techno-economic analysis

This Chapter has been published online as:

Clare, A., Gou, Y-G., Barnes, A., Shackley, S., Smallman, T.L., Wang, W., Jiang, D. & Li, J. (2015). *Should China subsidise cofiring to meet its 2020 bioenergy target? A spatio-techno-economic analysis.* Global Change Biology Bioenergy, DOI: 10.1111/gcbb.12264

Author Contributions

Abbie Clare: Formulated the concept for the paper, coordinated the collection of data, directed geographical analysis of the data in GIS, calculated the technical and financial parameters in R, wrote the paper.

Ya-Qing Gou: Updated the coal-fired power station data set provided by Jia Li (the nature of this job meant that it had to be completed by a fluent Chinese speaker), and assisted with geographical data analyses during a series of joint data analysis sessions

Andrew Barnes: Provided suggestions regarding financial analysis options

Simon Shackley: Provided overall guidance on paper concepts, and gave suggestions for the final draft of the manuscript. Thomas Luke Smallman: Provided trouble-shooting assistance for R coding

Wen Wang: Provided county-level data on crop yields for maize, rice and wheat

Dong Jiang: Provided land-use and province boundary data for China

Jia Li: Provided an early version of the coal-fired power stations data-set, which was subsequently updated by Ya-Qing Gou.

5.1 Chapter Rationale

The analyses of Chapters 3 and 4 demonstrate that biochar is unlikely to be an attractive technological option for Chinese farmers, businesses, or those interested in finding cost-effective carbon mitigation technologies for China's straw, particularly when the competing uses of biomass feedstocks for bioenergy are considered. In light of this finding, this chapter shifts the focus away from biochar as the primary use of interest for China's straw, towards bioenergy. Specifically, this chapter investigates China's existing bioenergy policy landscape, and addresses Objective 4 of the thesis by exploring how much bioenergy (TWh of electricity) could profitably be produced if China were to extend its feed-in-tariff for bioenergy to include low energy replacement ratio cofiring of agricultural residues in existing coal-fired power stations. This additional objective, although not focused on biochar, remains grounded within the overall framework of ecological modernisation, as it investigates whether there is an economic and environmental win-win solution for China's straw feedstocks that can be achieved through a change to their bioenergy policy design.

5.2 Introduction

Over the past 30 years, China has displayed an unprecedented average annual GDP growth rate of 9% (NBSC, 2012). During this time the nation's primary energy consumption and annual CO_2e emissions have both increased by over 550%, showing a close correlation (Figure 5.1; (EIA, 2015; World Bank, 2015)). In light of the evidence that energy consumption and greenhouse gas (GHG) emissions have been causally linked up to this point (Fei *et al.*, 2011; Wang *et al.*, 2011a; Li and Leung, 2012), China is now taking substantial steps to "de-couple" its energy use from CO_2e emissions, driven predominantly by environmental and climate change concerns (Li and Wang, 2012).

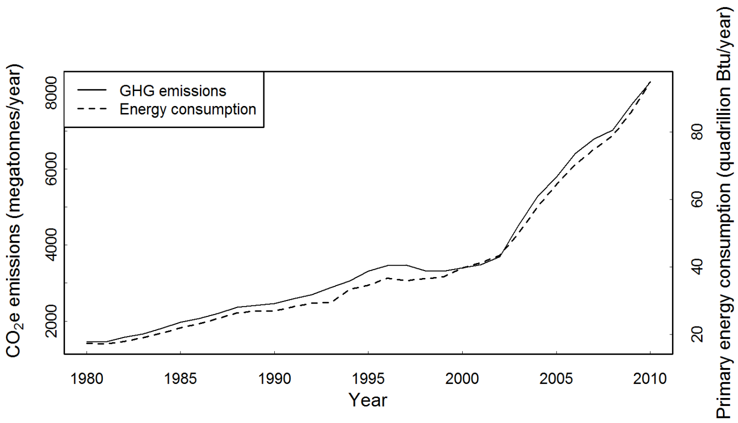


Figure 5.1: Graph of China's annual CO_2e emissions and primary energy consumption from 1980-2010

5.2.1 Renewable energy development in China

Decarbonisation of the energy sector is a central priority within the de-coupling goal, particularly because China's energy generation sector is dominated by coal-fired power stations, which produce 70% of the gross national energy supply (Li and Leung, 2012; Yuan *et al.*, 2014). Recognising this reliance on coal and the associated environmental implications, China has developed a variety of growth and emission trajectory targets, the latest of which is that the carbon intensity (measured as the energy-related CO_2e emissions per unit of GDP) of the economy must be reduced by 40-45% of 2005 levels by 2020 (Yuan *et al.*, 2014), with related targets for the share of non-fossil fuel energy to increase to 11.4% and 15% of the total energy supply, by 2015 and 2020, respectively *(ibid).* As part of this target, the State Council issued a plan in 2014 to cap coal consumption at 4.2Gt by 2020, and to reduce its share within the energy mix to 62% (Cornot-Gandolphe, 2014).

There has been substantial progress in meeting these renewable energy targets. By 2010 China had installed 216GW of hydro-power, 31GW wind power, 5.5GW biomass power and 0.8GW solar power, representing 113%, 620%, 100% and 160% of the original 2010 targets for each technology, respectively (Yuan *et al.*, 2014). However, despite having met the 2010 target for biomass energy installation, concern is growing over China's ability to meet the 2020 target of 30GW installed biomass power capacity. Meeting this target would produce an estimated annual 148.8TWh of bioenergy (Xingang *et al.*, 2013), which is equivalent to 3.1% of China's 2012 total net electricity generation of 4,768 TWh (EIA, 2015). Despite investing over $10 billion in biomass energy development between 2006 and 2011 (Xingang *et al.*, 2013), reports suggest that China's second largest biopower plant operator has not started construction on any biomass projects during 2012-2014, despite reporting a gross profit of $14.8m for its biomass projects in 2011, and having submitted plans for a further 26 biomass power plants (Gosens, 2015).

Various reasons have been cited for the slowdown in construction of biomass power plants, including high feedstock prices, poor coordination between projects and technical operating difficulties (Han *et al.*, 2008; Sang and Zhu, 2011; Zhao and Yan, 2012; Xingang *et al.*, 2013; Zhang *et al.*, 2013b; Yuan *et al.*, 2014). Moreover, there are reports that the existing financial support available through subsidies, grants and the renewable energy feed-in-tariff may not be sufficient to meet the 8% internal rate of return (IRR) that Chinese regulations outline as expected for investments in the power sector (Gosens, 2015).

5.2.2 Current support for bioenergy generation from crop residues

At present, China provides various capital grants, tax breaks and a feed-in-tariff (a flat rate of \$0.12 kWh^{-1}; (Zhang *et al.*, 2014)) to bioenergy projects that utilise agricultural residues to generate electricity. The targeting of agricultural residues is important, as China produces an annual 800 million tonnes (Mg) of straw, of which an estimated 505 Mt are available after retaining sufficient straw to maintain soil quality (Jiang *et al.*, 2012). However a significant proportion of this biomass resource is burned in-field as a waste, as a result of reduced demand for straw as a household fuel, a decline in the proportion of households keeping cattle, a scarcity of on-farm labour for straw collection, and the imperative for increasingly time-poor farmers to quickly dispose of waste residues before planting the next crop (Wu *et al.*, 2001; Lin and Song, 2002; Yu, 2003; Cao *et al.*, 2008; Rae, 2008). Despite the Chinese government announcing a variety of straw burning bans since the late 1990s, enforcement has proven difficult, costly and ineffective (Jingjing *et al.*, 2001; Qu *et al.*, 2012).

Instead, the government has established policies that financially incentivise the use of these residues as feedstock for bioenergy generation (Zhang *et al.*, 2014). However these subsidies are only available to units deriving 80% or more of their power from biomass. This restriction effectively rules out the cofiring of agricultural residues in existing coal-fired power stations, because cofiring biomass with high ash contents (such as agricultural residues) tends only to be technically feasible when approximately 10% of the coal feedstock is offset on an energy replacement basis (Al-Mansour and Zuwala, 2010; Tumuluru *et al.*, 2011; IEA, 2012), due to the risks of fouling and slagging associated with ash build up. This reticence to support cofiring at lower energy replacement ratios stems in part from perceived difficulties in verifying the ratio of cofired biomass at the coal fired power stations, and thus calculating the level of subsidy to award to each producer (Dong, 2012; Liu *et al.*, 2014b; Gosens, 2015). However there remains significant interest in the concept of cofiring in China, with a variety of scoping projects commissioned through international partnerships (DECC, 2008; Minchener, 2008), government-funded demonstration plants in Shandong and Shaanxi provinces (Liu *et al.*, 2014c) and various research studies reporting that cofiring agricultural residues in China can be both technically feasible and financially viable (Lu and Zhang, 2010a; Wang *et al.*, 2011b; Liu *et al.*, 2014c).

5.2.3 The benefits of cofiring

In theory, cofiring should lead to a variety of positive environmental outcomes. For example, Mann and Spath (2001) report that cofiring rates of 5% and 15% by heat input can reduce GHG emissions from a coal fired power plant by 5.4% and 18.2%, respectively. Moreover, cofiring can also reduce SO_2 and NO_x emissions (Mann and Spath, 2001; Huang *et al.*, 2006; Basu *et al.*, 2011), which are significant contributors to acid rain.

Cofiring is also significantly more desirable than biomass-only plants from the perspective of energy conversion: the average energy conversion efficiency of a biomass-only power unit is 25% (van Loo and Koppejan, 2008), whereas coal-fired powerstations are around 36% in OECD countries, increasing to 45% for more modern units (Wicks and Keay, 2005). Moreover, directly cofiring biomass with coal at relatively low ratios (5-10% energy equivalent) requires only minor alterations to the pre-existing feed and feedstock storage facilities, whereas building new bioenergy units entails construction, land, technical expertise, staffing and administrative costs (Zhang *et al.*, 2014).

With the growth of biomass-only units in China slowing or even reversing under the existing subsidy scheme, this paper therefore investigates how extending China's current bioenergy subsidy scheme to include cofiring could assist in meeting the country's target to install 30GW of bioenergy generation capacity by 2020. This question is examined through spatial, technical and financial lenses, using a unique geographically-linked dataset of China's coal-fired power stations and agricultural residue distribution.

Three previously unanswered questions are addressed. First, the geographic proximity of the three main agricultural residues (maize, wheat and rice straw) to China's existing coal-fired power plants is assessed, providing the first national-scale study of straw and power plant spatial co-location. Second, data on power station size, efficiency and technical cofiring capacity is combined with three straw removal scenarios, to assess the technical potential for cofiring across China. Finally, the financial case for agricultural residue cofiring from the perspective of investors is investigated. To date there has been suggestive case-study evidence that China's power stations can cofire straw profitably (Minchener, 2008; Lu and Zhang, 2010a), however this analysis is the first national scale estimate of the number of TWh that powerstations could produce from cofiring agricultural residues if the existing bioenergy subsidy scheme was extended to include cofiring.

5.3 Materials and Methods

5.3.1 Coal-fired power station data

China's coal fired power stations were identified using publicly available lists for 2006-2012 published by the China Electricity Council (2014) which provides power station names, administrative addresses and installed generating capacities (GW) of individual generation units. No power plants are located on China's islands, therefore this analysis focuses exclusively on mainland China.

The geographic coordinates of each power station were determined by searching for its administrative address online and associated instructions on how to visit the power plant by road. Google Earth software was then used to pinpoint exact geographic coordinates for the power plants, using the most recently available images. The x-y coordinates of the power stations cannot be published for security reasons. Overall, 268 powerstations were located, totalling 403GW of installed power generation capacity. Based on recent estimates, China's coal-fired generation capacity will be 960GW in 2015 (Industrial Efficiency Policy Database, 2014) and thus the dataset is able to account for 42% of this estimate. It is likely that many of the smaller powerstations on the lists could not be found due to China's recent policy of shutting power plants with low efficiency and poor environmental records (Zhang and Cheng, 2009; Chen and Xu, 2010). It is also likely that many larger, efficient (1GW+) plants have been built since the last available list was published (2012), which are also therefore not accounted for. Table 5.1[1] provides a summary of the number of power stations (n) and total installed capacity (GW) within geographical sub-regions of China. The listed regions are provinces, unless otherwise specified.

[1] a = Municipality, b = Autonomous region

Table 5.1: The number (n) of powerstations and total installed capacity (GW) of powerstations that were geographically located in each of China's sub-regions

Area	n	GW
Anhui	16	21
Chongqingshi[a]1	1	2
Fujian	7	16
Gansu	1	2
Guangdong	20	28
Guangxi[b]	4	5
Guizhou	7	11
Hebei	16	27
Heilongjiang	6	9
Henan	16	21
Hubei	9	12
Hunan	9	13
Jiangsu	32	46
Jiangxi	4	6
Jilin	3	4
Liaoning	7	11
Neimenggu[b]	15	27
Ningxia[b]	4	6
Shaanxi	11	15
Shandong	23	34
Shanghai[a]	11	16
Shanxi	20	27
Sichuan	4	6
Tianjin[a]	5	7
Xinjiang[b]	1	1
Yunnan	4	7
Zhejiang	12	23
TOTAL	268	403

Expert opinion and literature (Xiong *et al.*, 2009; Chen and Xu, 2010) were used to estimate the energy conversion efficiency of each power station, based on the size of individual units that make up the largest proportion of its total installed generation capacity (see Table 5.2). For example, a power station made up of 2 x 350MW units and 1 x 600MW units would be assigned to the 300-600MW category, whereas one of 1 x 350MW units and 1x600MW unit would be assigned to the 600-1000MW category. Where the proportional contributions are even, the powerstation is assigned to the higher unit capacity group.

Table 5.2: Estimated energy conversion efficiencies for power plants, based on the size of individual units that make up the largest proportion of the overall generation capacity

	<300MW	300-600MW	600-1000MW	>1000MW
Energy conversion efficiency (%)	30	34	39	45
No. of plants (n)	10	92	160	6

5.3.2 Agricultural residue data

County-level statistical data of grain yields was sourced from China's National Bureau of Statistics (NBS), generated from a 2006 county-level agricultural survey combined with agricultural census data (Wang *et al.*, 2013b). The data focuses on maize, wheat and rice yields, which together account for 81% of China's agricultural residue production (41%, 16% and 24% for maize, wheat and rice, respectively; (Jiang *et al.*, 2012)) and which are the most commonly used fuels in Chinese biopower plants (Gosens, 2015). We do not consider purpose-grown bioenergy crops as they are small in number, subject to strict land use limitations, geographically dispersed, and do not qualify for the same subsidies as bioenergy generated from agricultural residues. In contrast, agricultural straw is plentiful, consistently produced each year, and widely distributed throughout the country.

Data on maize, wheat and rice grain production was transformed into an estimate of straw energy potential using data on residue:crop ratios, straw moisture content and straw energy content (Cuiping *et al.*, 2004; Ming *et al.*, 2008). Table 5.3 details the assumptions made for each straw type. These are the same parameters as those used by Wang *et al.* (2013b).

Table 5.3: Assumptions in calculating available straw energy from county statistics of grain production

	Rice	Wheat	Maize
Residue:crop ratio	0.68	0.73	1.25
Moisture content (%)	15	15	15
Straw energy content (MJ kg^{-1})	14.66	16.56	16.64

The agricultural residue data was assigned to geographical units using a farmland distribution map at 1:100,000 scale, obtained from the Resources and Environmental Sciences Data Centre (RESDC) and the Chinese Academy of Sciences. Straw energy values (MJ) were assigned to each geographic unit (1000m x 1000m pixel) based on the area of farmland contained within each pixel, assuming that all rice is allocated to wet land, and that maize and wheat are allocated equally between dry and wet land:

$$P_{i,j,t} = \frac{L_{i,j,t}}{\sum L_{i,j,t}} \times E_{j,t}$$

where P = the energy (MJ) contained within pixel i, of county j, of land type t (wet vs. dry) , L = area of land contained within pixel i, of county j, of land type t, and E = the total energy (MJ) contained within county j, for land type t.

5.3.3 Sustainable rates of straw removal

A proportion of straw residues produced each season must be ploughed back into the soil in order for straw removal to be a sustainable practice that does not harm long-term soil quality and productivity (Lal, 2004). Appropriate straw retention rates vary significantly according to soil type, weather patterns and crop growth conditions. This variation is so high that some experts suggest that there can be no accepted universal minimum standard for crop residue retention, and that field or even sub-field level decisions are most appropriate (Karlen and Johnson, 2014). Therefore three straw removal rate scenarios are used, in order to reflect the high level of uncertainty around this model parameter:

Straw removal scenario 1: In order to calculate the technical potential of straw to produce bioenergy through cofiring, this scenario assumes that only the stubble remaining after harvest is returned to the field, and that all other crop residues are available for bioenergy production. This mirrors the assumptions of Wang *et al.* (2013b), who use coefficients that are calculated according to whether a crop is machine or hand harvested, and the resulting height of straw stubble (expressed as a proportion of total straw weight) that remains in the field. The harvest collection proportions for maize, wheat and rice are 0.95, 0.76 and 0.78, respectively (Ming *et al.*, 2008).

Straw removal scenario 2: In this scenario, the results of Jiang *et al.* (2012) are used to guide assumptions for the percentage retention of straw that is necessary to sustain soil fertility. Jiang *et al.* (2012) calculate that 505 Mt of a possible 800 Mt of straw in China are available for bioenergy production, after accounting for sufficient straw being returned for soil conservation purposes. This suggests that 37% of straws are retained, and 63% are available. After accounting for the stubble remaining in the soil (as per scenario 1), scenario 2 assumes that 37% of the remaining maize, wheat and rice straw is retained for soil fertility purposes, whilst 63% is available for bioenergy production.

Straw removal scenario 3: This scenario uses a conservative estimate of necessary straw retention, assuming that 50% of maize, wheat and rice straw must be retained, after accounting for the straw stubble left in the field. This fits well with information from a variety of studies on straw removal rates in China, which suggest that straw removal should be minimised to ensure ongoing soil productivity and health (Li *et al.*, 2006; Ming *et al.*, 2008; Wang *et al.*, 2010).

5.3.4 Competing uses of feedstocks

Although straw residues are commonly used in China for activities such as papermaking and animal forage, it was not possible to account for the local-level demand for these activities in a national scale model. However, the data sources for straw feedstock purchase prices used in the model (Zhao and Yan, 2012; Xingang *et al.*, 2013; Zhang *et al.*, 2013b; Gosens, 2015) are assumed to account for the effect of such competing uses on the price of feedstocks and therefore the estimated financial viability of cofiring. A range of straw prices are also used in the sensitivity analysis to test the impact of changing feedstock costs on bioenergy generation potential.

5.3.5 Straw collection radii and technical cofiring ratios

Agricultural wastes can be widely dispersed and difficult to collect, particularly within China's fragmented and small-scale farming system (Huang *et al.*, 2012b). Therefore the financially viable straw collection radius will vary for each powerstation, depending on local conditions. Research suggests that a wide range of radii are possible, from 20km (Minchener, 2008; Zhang *et al.*, 2013b) up to 40 or 50km (Liu and Huang, 2011; Thomas *et al.*, 2013; Liu *et al.*, 2014c). Given this level of uncertainty, technical cofiring potentials are presented for both 20km and 50km straw collection radii. Where collection radii overlap, the straw within the overlap area is evenly distributed between the powerstations whose collection radii are overlapping. This ensures that straw is not double-counted.

There is also uncertainty regarding the cofiring ratio of biomass to coal. This depends on two key issues. The first factor is the nature and chemical composition of the biomass being cofired. For example, cofiring wood with coal can achieve higher ratios than cofiring herbaceous biomass, because the ash content of wood is lower, and thus the potential for fouling and slagging of the coal-boiler is lower (Werkelin *et al.*, 2010; IEA, 2012; Teixeira *et al.*, 2012). The second factor relates to the method of cofiring. This can either be direct (where biomass is sent through the same pulverisation process as coal and directly fired within the same boiler), indirect (where a biomass gasifier converts solid biomass into a fuel gas, which can be cleaned and then burned in the coal boiler furnace) or parallel (where a completely separate biomass boiler is installed and the steam produced is utilised in the coal power plant system; (Al-Mansour and Zuwala, 2010)). Direct cofiring requires the fewest modifications and thus the least additional capital investment, however it also facilitates the lowest ratio of biomass:coal cofiring compared to indirect and parallel cofiring configurations. In China, where cofiring is a nascent concept, it is most likely that direct cofiring will be used, and this is therefore the assumed technology for analysis.

5.3.6 Financial assessment of cofiring

The internal rate of return (IRR) is calculated for individual power plants (at 2015 prices, adjusted using Inflation EU (2015) and assuming a currency conversion rate of 6.14 renminbi to 1 US dollar) in order to determine the financial viability of cofiring to investors if China were to extend the current bioenergy policy to include cofiring. In order to calculate individual IRRs, the maximum cofiring rate of each powerplant is calculated according to its installed capacity (GW), energy conversion efficiency (see Table 5.2), annual operating time (5694 hours per year; (Gosens, 2015)) and MJ of straw residues available within a 20km or 50km radius. A 10% transport and handling loss is also accounted for during straw collection. The upper limit for cofiring is assumed to be 10% coal energy replacement with agricultural residues, due to the assumption that direct cofiring will be the technology used, which means that cofiring rates must remain relatively low to avoid boiler fouling. Cofiring is also assumed to reduce powerstation efficiency by 1% (Minchener, 2008; Wang *et al.*, 2011b), the financial loss from which is included in the IRR calculation. The IRR is calculated only for the biomass cofiring element, and thus represents the additional returns that a power plant might expect when choosing to cofire a biomass:coal ratio appropriate to its straw availability, as compared to the status quo of firing coal only. The IRR for each powerstation is calculated over a ten year period, and assessments are conducted before taxes.

5.3.7 Financial parameters

Literature estimates of the capital costs of converting coal-fired power stations to cofiring capability vary from zero costs (where biomass is briquetted before cofiring; Liu *et al.* (2014c)) to between \$59-426 kW^{-1} installed biomass capacity (US Dept. of Energy, 2000; Al-Mansour and Zuwala, 2010). The baseline assumption of this study is therefore the mid-point of this latter range (\$243kW^{-1} installed biomass capacity), and this figure is tested in the sensitivity analysis.

Coal price is assumed to be \$97Mg^{-1} and coal energy density is assumed to be 23,000 MJ Mg^{-1} (Bloomberg, 2014; Liaoning Government, 2014). Average straw energy density is calculated individually for each powerstation collection radius according to the proportion of maize, wheat and rice straw, and the energy densities of these straw types (see Table 5.3). Straw price is assumed to be \$47Mg^{-1}, calculated as a middle range estimate from recent publications regarding the production of bioenergy from agricultural residues in China (Zhang *et al.*, 2013b, 2014; Liu *et al.*, 2014c; Gosens, 2015).

Costs of straw transportation and pre-treatment were derived from a number of sources. According to Liu *et al.* (2014c) and Zhang *et al.* (2013b), straw is collected and briquetted by a "middle-man" enterprise, which then sells briquettes to the power station. These pre-treatment costs are estimated at \$29 Mg^{-1} plus a 10% profit for the straw briquette business of \$2.9 Mg^{-1}.

Straw transportation was assumed to be by road. Although the rail network is a major transport means for coal to China's powerstations, straw resources are low in energy density and far more dispersed at their source than coal. Therefore they are better accessed by road than by rail. Straw transportation distance was determined for each powerstation using an equation from French (1960) assuming a circular radius and square road grid:

$$D_i = \sqrt{\frac{S_i}{640 \cdot Y_i \cdot d_i}}$$

where D_i is the average distance (miles) each Mg of straw feedstock is hauled for power station i; S_i is the annual amount of feedstock required for power plant i, multiplied by 0.5 to reflect two growing seasons; Y_i is the average biomass yield per acre in the 50km collection radius of power station i; d_i is the fraction, or density, of land on which agricultural residues are produced within the 50km collection radius of each power station i; and 640 is a conversion factor for the number of acres per square mile. The mean calculated distance per Mg of straw was 86km, with a range of 24km to 168km.

Coal-fired power plants generate some revenue from sales of fly-ash to cement industries. Research has demonstrated that cofiring biomass with coal at up to 25% energy replacement ratios does not significantly affect fly-ash quality and is able to meet the Chinese standard (GB/ T1596-2005) for sale to the cement industries (Wang *et al.*, 2011b). However, cofiring biomass with coal may reduce the quantity of fly-ash produced per unit energy output, as biomass contains a lower proportion of ash per unit weight than coal (Al-Mansour and Zuwala, 2010). Therefore a conservative assumption is made that sales of fly-ash at each power plant decrease linearly with the ratio of biomass cofiring. I.e., a 3% rate of cofiring would lead to a 3% reduction in revenue from sales of fly-ash. Fly-ash is assumed to be sold at \$6.5Mg^{-1} and it is assumed that, under standard operating conditions, 100 Mg coal would produce 3Mg fly-ash (expert opinion).

The grid-purchase price for electricity generated from coal varies according to the contracts agreed between power station owners and the Chinese government, which are based on powerstation age, efficiency and sulphur emissions. Expert opinion and available data (Bloomberg, 2012; Gosens, 2015) suggest that the average price is around \$0.068kWh^{-1}, and under China's bioenergy subsidy scheme, this price increases to \$0.12kWh^{-1} for bioenergy derived from agricultural residues (Zhang *et al.*, 2014).[2]

5.4 Results

5.4.1 Geographic co-location of China's agricultural residues and coal-fired power plants

There is substantial co-location of straw energy (terajoules; TJ) and existing coal-fired power stations in China. Figure 5.2 provides a visual depiction of the distribution of China's straw resources, overlaid with the location of power plants, and their respective 50km straw collection radii.

Table 5.4 outlines the energy (TJ) contained in the maize, wheat and rice straw resources that are located within 20km and 50km radii of the powerstations, and the proportion of China's total maize, wheat and rice straw resources that this accounts for. Our dataset estimates China's maize, wheat and rice straw resources at 6,100,000 TJ, which is broadly comparable to other estimates of 5,861,000 TJ (Jiang *et al.*, 2012) and 4,390,000 TJ (Wang *et al.*, 2013b). Notably, 39% of China's straw resources are situated within 50km of the powerstations identified

[2] A step-by-step explanation of the stages undertaken for this analysis is available in Appendix 3.

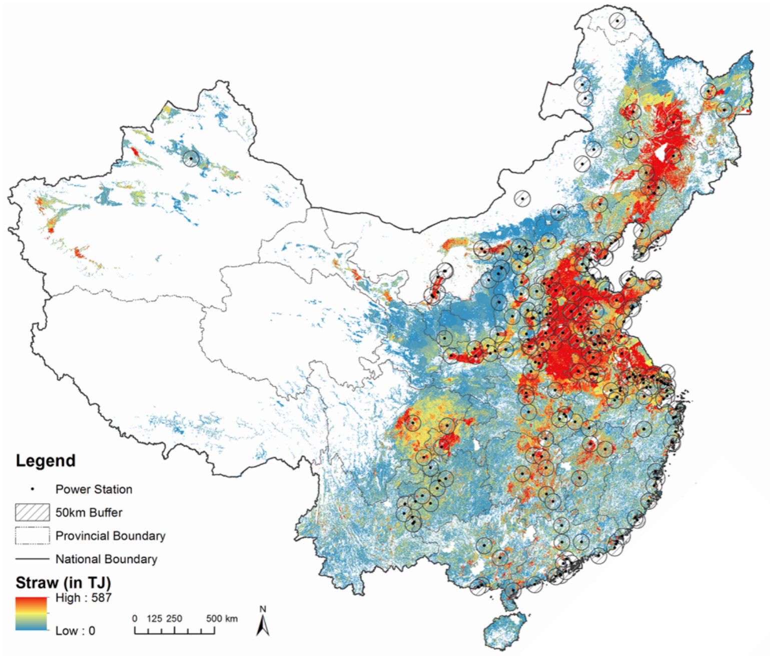


Figure 5.2: Geographic co-location of China's straw resources (TJ per km^2) and power plant collection radii (50km)

in the dataset.

Table 5.4: Straw availability (TJ and % of total) within 20km and 50km radii of powerstations

	Straw (TJ)	Of total (%)
20km	570,000	9
50km	2,370,000	39

5.4.2 Technical cofiring potential of power plants

Table 5.5 outlines the number of powerstations that can cofire at a range of energy replacement ratios (1-10%) at 20km and 50km radii, and the estimated number of TWh that would be produced. We find that 68, 64, and 59% of powerstations can cofire at 1% or more using straw within a 20km radius for straw removal scenarios 1, 2 and 3 respectively, and that 81, 81, and 78% of powerstations can cofire at 1% or more within a 50km radius, for straw removal scenarios 1, 2 and 3 respectively. Interstingly, the majority of cofiring potential occurs at low cofiring rates, both in terms of the number of powerstations able to cofire at a given rate and the TWh that they produce.

Combining this data into a cumulative analysis, Figure 5.3 demonstrates that, if all power plants were to cofire their highest spatially and technically feasible straw:coal ratio, up to a maximum of 10%, China could produce an annual 45, 35, or 27TWh of bioenergy for straw scenarios 1, 2 and 3 within a 20km straw collection radius, or 117, 102, or 89TWh of bioenergy for straw scenarios 1, 2 and 3 within a 50km straw collection radius.

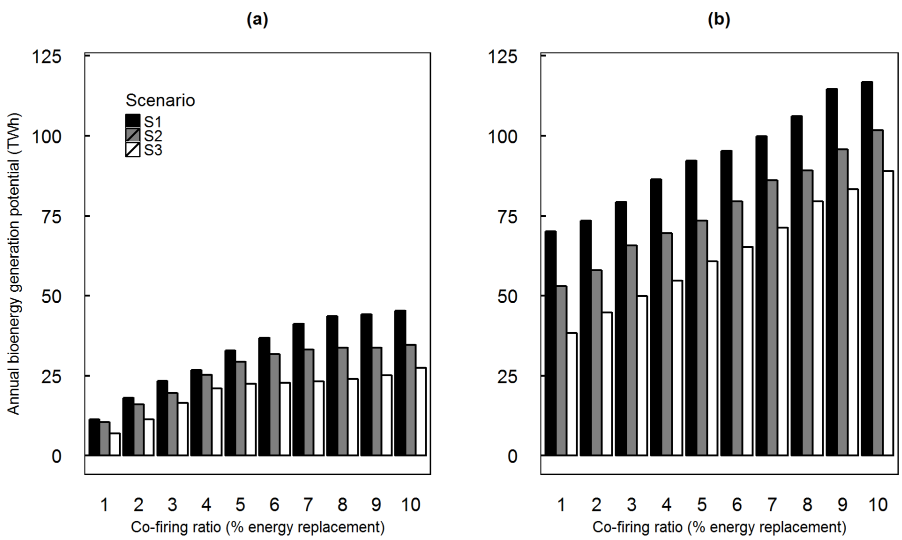


Figure 5.3: Cumulative totals of annual bioenergy generation (TWh) from agricultural residue cofiring at 1-10% cofiring ratios, within 20km (panel a) and 50km (panel b) straw collection radii, and under three straw removal scenarios (S1-S3)

These numbers are significant when compared to the current generation totals of

Table 5.5: Number of powerstations (n) that can co-fire at each ratio, and the subsequent bioenergy that this would produce (TWh) in each of the three straw collection scenarios

Radius	Straw Scenario		1%	2%	3%	4%	5%	6%	7%	8%	9%	10%	Total
20km	1	n	66	44	21	11	14	9	9	4	1	2	181
	1	TWh	11.3	6.8	5.3	3.3	6.2	4	4.3	2.3	0.7	1.2	45.3
	2	n	79	36	16	17	12	6	3	1	0	1	171
	2	TWh	10.4	5.6	3.5	5.7	4.1	2.4	1.4	0.6	0	0.9	34.5
	3	n	81	28	20	16	4	1	1	1	2	3	157
	3	TWh	7	4.4	5.1	4.6	1.4	0.3	0.5	0.7	1.2	2.3	27.4
50km	1	n	108	19	21	17	14	7	8	9	12	3	218
	1	TWh	70.1	3.3	5.9	7	5.9	3.1	4.6	6.3	8.5	2.1	116.8
	2	n	95	27	26	12	10	12	12	5	10	7	216
	2	TWh	53	5	7.7	3.8	4	5.9	6.7	3.1	6.6	5.9	101.7
	3	n	79	34	21	15	15	10	11	11	5	8	209
	3	TWh	38.3	6.5	5.2	4.7	6	4.6	6	8.2	3.8	5.8	89

other renewable energy technologies in China. For example, solar installations in China produced 8.7TWh in 2013, wind produced 141TWh, and nuclear contributed 112TWh. Moreover, these results suggest that cofiring straw could contribute significantly to China's 2020 target to install 30GW of bioenergy generation capacity, which is equivalent to a generating capacity of 148.8TWh per year (Xingang *et al.*, 2013). Within a 50km radius, the technical potential for cofiring is estimated at 78%, 68% and 60% of this target for straw removal scenarios 1, 2 and 3, respectively.

5.4.3 Economic feasibility of cofiring with and without subsidy support

The internal rate of return (IRR) is calculated for the powerstations that are able to cofire at a biomass:coal energy replacement ratio of above 1% (up to a maximum of 10%), within a straw collection radius of 50km, under each straw collection scenario. The TWh of bioenergy produced by all cofiring power plants with IRRs of 8% and over are summed together to estimate the bioenergy generation that would result under a variety of technical and financial scenarios.

Under baseline assumptions (see Table 5.6[3]) and without the support of China's bioenergy feed-in-tariff, cofiring makes a significant loss at all power stations and zero TWh of bioenergy are produced.

In contrast, if the current bioenergy feed-in-tariff (\$0.12 kWh^{-1}) is used to value the bioenergy produced from agricultural residues at the coal-fired power stations, cofiring is profitable at 131, 119 and 100 powerstations across China, generating

[3] a = In the absence of appropriate range data, a mid-range value is taken from the literature and varied by +/- 50%

Table 5.6: Parameter baseline, lower and upper values used for sensitivity analysis

Parameter	Baseline	Low	High	References
Capital cost ($ kW installed capacity^{-1})	243	59	426	US Dept. of Energy, (2000); Al-Mansour & Zuwala, (2010)
Straw purchase (from farmer; $ Mg^{-1})	47	34	78	Zhang et al., (2013, 2014b); Liu et al., (2014)
Straw pre-treatment[a] ($ Mg^{-1})	32	16	48	Liu et al., (2014); Zhang et al., (2013)
Straw transport[a] ($ Mg^{-1} km^{-1})	0.49	0.24	0.73	Liu et al., (2014)
Coal purchase[a] (at power plant gate; $ Mg^{-1})	97	48	145	Bloomberg, (2014)
Coal energy density (MJ Mg^{-1})	23000	18700	29300	Bloomberg, (2014); Liu et al., (2014); Liaoning Government, (2014);
Coal energy price ($ kWh^{-1})	0.068	0.043	0.08	Bloomberg, (2012); Gosens, (2015)

a cumulative total of 91.5, 75.9 and 62.2 TWh of bioenergy under straw scenarios 1, 2 and 3, respectively. This represents 62, 51 and 42% of China's expected bioenergy generation under the 30GW target, equating to between 1.3 and 1.9% of China's 2012 total electricity net generation (EIA, 2015).

Sensitivity analysis on key economic and energetic parameters using values from Table 5.6 demonstrates that the profitability of cofiring, and resultant anticipated TWh of bioenergy generation, is strongly influenced by a variety of parameters.

Figure 5.4 shows that the changes in the purchase prices of coal (Buy Coal) and straw (Buy Str) have the greatest impact on the profitability of, and related predicted energy generation from, cofiring agricultural residues in China's powerstations. The straw transportation (Tran Str), straw treatment price (Str Trt) and the grid price for coal-fuelled electricity (Coal Elec) are also important in determining cofiring profitability, whereas the energy content of coal (Coal En) and the capital costs of retrofitting powerstations to accept straw biomass (Cap) have a relatively small impact on the anticipated bioenergy generation as it relates to profitability. In the third and most conservative straw removal scenario, the upper limits of straw cost (\$78 Mg^{-1}) and lower limits of coal cost (\$48 Mg^{-1}) result in the estimate of bioenergy production from cofiring dropping to under 10 TWh, representing 6-7% of China's 30GW target.

5.5 Discussion

Overall there is significant spatio-techno-economic potential for China to generate sizeable quantities of bioenergy by cofiring available agricultural residues

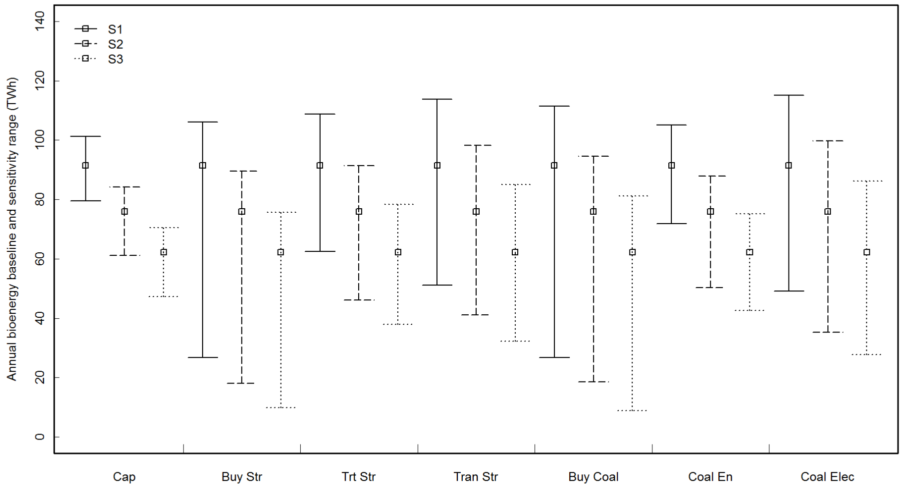


Figure 5.4: Variation in annual bioenergy generation (TWh) from cofiring according to sensitivity analysis using key financial and energetic parameters for the three straw removal scenarios (S1-S3)

in its coal-fired power stations, if the government extends the current bioenergy feed-in-tariff to include low ratio biomass cofiring operations. This indicates that the suggested change to the existing bioenergy subsidy system may indeed be an economic and environmental win-win opportunity for China's straw resources.

Under baseline economic conditions, and without subsidy support, cofiring agricultural residues is not profitable for powerstations. However, this study suggests that extending the subsidy to include cofiring could stimulate between 62-92 TWh of bioenergy generation, depending on the assumed removal rate of straw. This could account for between 42-62% of the bioenergy generation that would be expected under China's 2020 target to install 30GW of bioenergy production capacity.

These results are subject to two caveats. Firstly, the profitability, and related bioenergy generation potential, of cofiring is highly sensitive to the purchase price of coal and straw. When varied independently, neither the highest straw price (\$78 Mg^{-1}) nor the lowest coal price (\$48Mg^{-1}) bring the estimated bioenergy generation to zero TWh, with a predicted output of 9.8TWh and 8.8TWh, respectively, under the most conservative straw availability assumptions (scenario 3). Nevertheless, if the straw price was to rise and coal price were to simultaneously fall, this could seriously affect the profitability of cofiring agricultural residues. However, one benefit to cofiring in comparison to biomass-only generation units is that cofiring operations are better able to respond to such changes in market conditions. For example, when the straw price is high, powerstations can focus on coal-fuelled electricity generation, and vice versa, without experiencing prolonged periods of reduced income. In contrast, biomass-only units are very vulnerable to changes in straw purchase price, and may suffer long periods of financial losses that can be hard to recover from. This may at least partly explain the reports of bioenergy plant shut-downs across China in recent years (Han *et al.*, 2008; Zhao and Yan, 2012; Xingang *et al.*, 2013; Zhang *et al.*, 2013b). Therefore the concerns over the sensitivity of these results to straw and coal purchase prices are important, but are arguably less significant in their impact on bioenergy generation from cofiring as compared to biomass-only bioenergy projects.

A second caveat is that China's 30GW target for installed bioenergy generating capacity is likely to be driven partly by a desire for additional electricity generating capacity, whereas cofiring works within existing installed capacity, directly replacing coal feedstock with biomass. Nevertheless, given the current challenges faced by biomass-only electricity generation units, it is possible that cofiring straw is a more efficient use of China's agricultural straw resources, and that total installed renewable generation capacity may more cost-effectively be expanded via solar, wind or hydro projects, rather than biomass-only units.

5.5.1 Conclusions

Overall, this analysis demonstrates that significant quantities of bioenergy can be generated by cofiring agricultural residues in China's existing coal-fired power stations, taking into account spatial, technological and financial opportunities and constraints. Moreover, this may be an underestimate of China's true cofiring potential, as the powerstation dataset used for this analysis is only able to geographically locate 42% of China's estimated coal-fired generation capacity for 2015 (Industrial Efficiency Policy Database, 2014). Given reports of the difficulties that biomass-only power generation units have encountered, and the relatively smaller investment costs, risks and vulnerability to biomass prices of cofiring compared to biomass-only operations, these results provide a convincing case for the Chinese government to extend their existing bioenergy feed-in-tariff to include cofiring operations at low biomass:coal ratios.

The following chapter discusses the potential of this apparent win-win solution in light of broader contextual factors affecting China's bioenergy policy and energy system. The chapter will also return to the idea of biochar as a win-win solution for agriculture and climate change, and consider the circumstances under which biochar may offer a more convincing contribution to improved environmental management and economic growth.

Acknowledgements

This work was funded by the EPSRC (Science and Innovation Award) and Scotland's Rural College (SRUC).

Chapter 6

Discussion and Conclusions

Throughout this thesis I have been exploring the win-win potential of converting China's straw feedstocks to biochar, and subsequently bioenergy, from the perspective of ecological modernisation. Similar to the theory of sustainable development, a central assumption of ecological modernisation is that economic growth can continue alongside improved environmental management, and key to this proposition is the idea that technological and/or policy solutions exist that can maintain growth and simultaneously contribute to ecological goals. In the case of biochar, the dominant win-win characteristics identified by research have been its role as an agronomic input (contribuitng to economic growth by increasing crop yields and/or improving the output efficiency of agricultural systems) and its potential as a climate change mitigating technology (by sequestering carbon within its structure, and delaying the release of biomass carbon back to the atmosphere for hundreds to thousands of years). With a predominance of physical science data to support these claims, but very little complementary socio-economic analyses available, the aim for this thesis was to generate applied, policy-relevant socio-economic analysis of biochar's win-win potential, placing the research questions within the context of China's agricultural systems and straw resources.

This concluding chapter summarises the win-win solutions that have been identified through this research, and discusses how they fit within the goals of ecological modernisation and, more broadly, how they might contribute towards achieving sustainable development.

6.1 Win-win solutions for bioenergy

Overall the results from this thesis suggest that win-win outcomes are most likely when China's straw resources are deployed for bioenergy generation, rather than for biochar production. The results from Chapter 3 cast doubt over the apparent economic attractiveness and poverty alleviation potential of biochar for smallholder farmers in China (thesis objective 1), whilst the results from Chapter 4 also demonstrate that biochar production from China's straw resources are not profitable for business investors, either in absolute terms (returning a negative NPV under every subsidy scenario) or in relative terms (returning a lower NPV than using straw for briquetting or gasification under every subsidy scenario; thesis objective 2). Thus, the agrononic/economic "win" of biochar appears insufficient for large-scale adoption of this technology in China.

Moreover, this research has also called the climate change mitigation/environmental "win" of biochar into question. Despite providing some carbon sequestration services compared to the baseline uses of China's straw (sequestering 1.06MgCO_2e per oven dry tonne), Chapter 4 also demonstrates that biochar production has lower carbon sequestration potential per tonne of feedstock processed as compared to briquetting and gasification. Moreover, it is also the least cost-effective climate change mitigation technology (measured as the subsidy required per tonne of CO_2e mitigated in order for an investor to break-even; thesis objective 3). Thus, the results from Chapters 3 and 4 strongly suggest that ecological modernisation win-win solutions are best achieved by using China's straw feedstocks for bioenergy generation, rather than biochar production.

Chapter 5 explores this apparent dominance of bioenergy further, and investigates possible win-win outcomes from a small change to China's bioenergy policy: the extension of the existing bioenergy feed-in-tariff to include low energy replacement ratio cofiring in existing coal-fired power stations (thesis objective 4). Again, there appear to be win-win outcomes available from this policy change, as the model suggests that China could generate the equivalent of 42-62% of its 2020 bioenergy installation target. This offers a commercial win for businesses looking to invest in cofiring, a governmental win for meeting China's national bioenergy capacity installation target, and environmental wins where this leads to an acceleration of coal energy replacement with biomass energy, improved efficiency of energy conversion for straw resources, and a reduction in harmful emissions from coal-fired power stations (Mann and Spath, 2001).

However, although these win-win solutions are attractive in theory, in practice the operationalisation and attainment of these wins can be difficult. The following section provides a critical discussion of the theory and application of win-win solutions.

6.2 Bioenergy in China: a true win-win?

The results from Chapters 4 and 5 present three apparent win-win solutions for China's straw. In Chapter 4, both briquetting straw for heat energy and gasifying straw for electricial energy appear to generate economic gains for investors and environmental gains from a climate change perspective. However, these results are subject to caveats. For example, briquetting has significant climate change mitigation potential predominatly because it is replacing coal that is being burned in inefficient boilers. As communities modernise and move towards more advanced heating fuels, straw briquettes may have much less climate mitigation potential. Moreover, although gasification appears to be a significant win-win for both economic and environmental aims in this analysis, there is growing recognition of the serious technical and financial difficulties that many of these gasification units have had in China (Han *et al.*, 2008; Sang and Zhu, 2011; Zhao and Yan, 2012; Xingang *et al.*, 2013; Zhang *et al.*, 2013b; Yuan *et al.*, 2014). These arise from issues with feedstock provision, poor project coordination and operational difficulties, none of which are taken into account in the win-win framework of assessment that has been used here.

Likewise, the win-win solution identified for cofiring in Chapter 5 can be critiqued from a similar perspective. Despite finding that China could generate the equivalent of 42-62% of its 2020 bioenergy installation target from a small change to its bioenergy policy design, it is unlikely that this apparent triple win will be straightforward to operationalise in practice. Firstly, the Chinese government are aware of the benefits of cofiring in comparison to biomass-only units, having collaborated with the EU on a cofiring scoping project (Minchener, 2008). However, they remain wary of providing financial support to the concept, due to fears over their inability to verify the volumes of energy that are produced from biomass in the powerstations, rather than from coal (Dong, 2012; Gosens, 2015). This is an understandable concern given China's struggle with corruption, and is one that may not be allayed by the findings in Chapter 5.

Moreover, within the broader context of global warming, this modest tweak to China's bioenergy policy is a very small contribution to mitigating climate change. It does nothing to address the underlying fossil-fuel dominated energy generation industry in China and may even arguably perpetuate carbon lock-in if coal-fired power plants are made slightly less carbon intensive and thus more justifiable from an environmental stand-point. There is also no guarantee that the coal offset through cofiring will not be burned in other powerstations without cofiring, making any calculations of climate mitigation impacts from this policy change highly uncertain. Finally, a nation-wide policy that facilitates businesses to profit from the removal of agricultural straw from China's fields could generate further environmental problems as a result of increased soil erosion and/or reduced soil organic matter levels, where straw is removed from the soil at unsustainable rates. Although the analysis in Chapter 5 accounts for three straw removal scenarios, in reality this is a farmer-level decision and, with straw already being treated as a waste by many farmers, the arrival of businesses offering payment for straw resources could exacerbate soil fertility problems in China. Given the potential for negative environmental consequences, it may therefore be more prudent for the Chinese government to focus on alternative renewable sources such as wind and solar energy, the bioenergy technical generation potential of which both greatly exceed bioenergy from China's biomass sources (Yuan *et al.*, 2012), and which are less likely to have negative consequences for the food production system.

From this perspective, the neatly packaged win-win solutions that are presented in Chapters 4 and 5 seem less feasible and more complex than the results initially suggest. In fact, it seems that apparent win-win solutions are often couched in highly complex contexts, the costs and benefits of which can often fall outside the relatively narrow framework that a win-win approach implies. If this is the case, the concept of win-wins may itself be flawed. The following section therefore critically discusses the win-win approach to environmental management and the extent to which this approach may be helping or hindering progress towards sustainable development.

6.2.1 A critique of the win-win concept

As previously mentioned in Chapter 1, the concept of sustainable development and the idea that economic growth can co-exist with environmental protection evolved in part in response to the *"apocalyptic horizons of environmental concern"* (Dryzek (2013), pg. 145) that were put forward in the 1970s by radical environmental groups, arguing that economic development and population growth would need to be restrained in order to stay within global environmental limits.

With rising popularity of the sustainable development concept and related theories like ecological modernization, the search for win-win solutions has intensified and become a key political narrative in high level talks on climate change and environmental management. In part this may be because sustainable development, ecological modernisation and the concept of win-wins are *"discourse(s) of reassurance"* (Dryzek (2013), pg. 175), suggesting that no hard choices will need to be made between economic growth and environmental management. Indeed, the theory of ecological modernisation is often described along a continuum of operationalisation, from "weak" to "strong", as first described by Christoff (1996) and since built upon by other scholars to describe their perceived and intended applications of the theory (Blowers, 1997; Berger, 2001; Toke, 2002).

Christoff (1996) describes "strong" ecological modernization as consisting of broad-ranging changes to society's institutional structure and economic systems, alongside demoncratic and participatory involvement of citizens, and the competent communication of environmental affairs. In contrast, "weak" versions of ecological modernisation emphasise narrow, technological solutions to environmental problems, imposed by a scientific and/or political elite. In light of these very different interpretations of the same theory, many scholars worry that weak ecological modernisation may be deployed in place of stronger versions, with the danger that this may *"legitimize the continuing instrumental domination and destruction of the environment"* Christoff (1996); pg. 497). Or, as Jänicke (2008) puts it, that there is a *"danger that we content ourselves with the "low hanging fruits" of marketable "win-win solutions" rather than tackling the larger, structural causes of environmental degradation"*. Arguably, this has occured in countries such as the US or UK, who have been slow to adopt an integrated policy approach to environmental issues, whereas countries such as Germany, Norway and Japan have made significant progress in this regard (Dryzek, 2013).

Related to the weak-strong continuum, Hajer and Versteeg (2005) recommend following a path of "reflexive ecological modernisation", which assumes that modernisation will continue but which encourages political and economic development to proceed on the basis of a critical self-awareness, where the qualitative nature and direction of the progress path is constantly evaluated and re-assessed. In short, many theorists are concerned that the apparent simplicity of ecological modernisation's win-win approach may be used without due care and attention, and a literature that is critical of ecological modernization, and the win-win concept in particular, has emerged around these concerns.

Evidence of this critical literature is particularly prominent in the field of international development, where the search for environment-development win-wins has become commonplace is recent years, with far-reaching implications. For example, projects are much more likely to receive institutional funding where numerous complementary (win-win) outcomes are expected under a neatly-packaged project proposal (Simon *et al.*, 2012). These outcomes often span health, livelihood, carbon and biodiversity benefits that are said to accrue to a broad range of actors, both local and global. However, in reality these projects often struggle to deliver on these win-win promises (Campbell, 2009; Phelps *et al.*, 2012; Simon *et al.*, 2012). For example, Tallis *et al.* (2008) found that only 16% of World Bank-funded projects claiming to deliver environment-development wins actually made major progress in both areas, suggesting that conceptualising win-wins on paper is much easier than achieving them in practice.

Moreover, critique of win-win solutions extends beyond simply failing to meet ambitious goals. An even more damaging consequence may be the dominance of win-win expectations discouraging project planners and participants from addressing the inevitable trade-offs that arise when implementing a policy or project of any complexity (Campbell, 2009; McShane *et al.*, 2011). In fact, there is often a benefit to explicitly acknowledging trade-offs and in bringing disparate parties together to negotiate, thus improving understanding between parties with different perspectives and creating appropriate compromise solutions (McShane *et al.*, 2011). Additionally, there are suggestions that the strength and apparent elegance of the win-win concept, and associated “pro-poor” technologies, may simply act to reinforce the power of “modern” or “western” technologies over local perspectives (Grieve, 2004), acting as a rhetorical tool to placate local concerns in order to advance a westernised agenda of development (Simon *et al.*, 2012). This raises the question of who defines what a win is, who it is for, and over what time frame. In fact, addressing these questions often exposes a deeply complex web of compromises and trade-offs that must be negotiated, which again raises the broader issue of how to facilitate effective participatory processes that ensure that all stakeholder views are fairly represented and accounted for. This is an incredibly complicated task, but this participation of multiple actors, and the inclusion of civil society alongside more formalised scientific and political institutions, is a defining feature of strong and reflexive versions of ecological modernisation (Christoff, 1996; Hajer and Versteeg, 2005).

In light of this discussion, it seems that the cofiring win-win outlined in Chapter 5 is a rather weak/non-reflexive version of ecological modernization. The policy change would be thought up and implemented by scientific, commercial and political actors, for whom there are short-term economic wins (investing in cofiring and profiting from available subsidies). Whilst there would also be short-term financial wins for farmers who are encouraged to sell their straw, they may suffer longer term losses as their soil fertility declines. If legislation was passed to ensure that soil health was not harmed through the removal of crop straws for bioenergy, this would make the support of cofiring a stronger version of ecological modernisation. However this too would be extremely difficult to monitor. In reality, removing large quantities of crop straws for what is, relatively speaking, a very small contribution to China's energy demands may end up creating more damage than good over time.

Overall it seems that a win-win approach to ecological modernisation can result in a rather narrow frame of reference within which to assess the potential contribution of a new technology and/or policy. This is certainly the case within this thesis, as biochar's potential to contribute to the process of ecological modernisation has been assessed on financial and climate change mitigation indicators only, albeit from the perspectives of a variety of stakeholders. As such, the following section considers the possibility of moving biochar away from the win-win discourse, into a more holistic framework of assessing its contributions to sustainable development.

6.3 Biochar: moving beyond win-wins

The results from Chapters 3 and 4 suggest that biochar will struggle to achieve its much-discussed win-win potential for agronomic impacts and climate change mitigation. Specifically, Chapter 3 suggests that biochar's proposed win of agronomic (and associated economic and/or poverty alleviation) benefits to farmers is unlikley to be realised using current biochar application models. The costs of biochar sourcing (either through commercial purchase or on-farm production) and the labour intensity of biochar application are rarely outweighed by the value of agronomic gains (increases in yield and inorganic fertiliser savings). As such, it is unlikely that biochar will offer an attractive livelihood option for farmers. Moreover, Chapter 4 suggests that biochar is not a commerically viable option for businesses in comparison to using straw for bioenergy generation, nor is it an environmental win in terms of relative climate change mitigation per tonne of feedstock processed, or in terms of its cost-effectiveness in reducing CO_2e emissions, compared to using China's straw feedstocks for bioenergy production.

Thus, it could be logical to conclude that biochar cannot live up to the win-win claims that have been made for its environmental and economic benefits. However, reflecting upon the previously discussed critiques of the win-win approach, it is possible that such a conclusion would reflect the narrow framework of the analytical approach used in this thesis, rather than biochar's actual potential to contribute to the process of sustainable development. For example, Chapter 3 concludes that biochar may not have potential as a poverty alleviation win, because it is not profitable for smallholders in China. However, poverty is about more than basic income, and also relates to the risks faced in the consistency of crop yields and/or the resilience of an agro-ecosystem to external events such as climate change or extreme weather. It remains possible that regular biochar application could reduce the variability of crop yields, or the vulnerability of an agro-ecosystem to external events (McHenry, 2009; Joseph, 2009), particularly where biochar can boost the soil fertility of degraded, fragile soils. Therefore, although Chapter 3 finds little evidence to support the idea that biochar can increase smallholder income, it would be premature to dismiss it entirely from the discussion of sustainable development.

Similarly, Chapter 4 finds that biochar cannot compete with bioenergy on either environmental or economic grounds for the use of China's straw feedstocks. However, this analysis does not consider or value factors such as improved soil fertility, reduced nitrate run-off pollution and/or the possibility that biochar could contribute towards remediation of salinised or heavy mental-contaminated land (Barrow, 2012; Lashari *et al.*, 2013; Zhang *et al.*, 2013d). Given China's high ratio of people to land, the increasing severity of its land contamination/degradation (Chen, 2007; Khan *et al.*, 2008; Bian *et al.*, 2013), and the population's growing preference for resource intensive foods such as meat and dairy, it is possible that technologies like biochar will become a more attractive use for China's straw resources, particularly if few other technological alternatives exist to achieve China's desired aims in this area. In contrast, as already mentioned, there are many alternative sources of renewable energy in China that have greater technical potential for energy generation (Yuan *et al.*, 2012) and that may be more efficient and cost-effective than bioenergy from straw feedstocks. Therefore the real determinant of wide-spread biochar adoption in China is likely to be the extent to which govermental priorities focus on the environmental services that biochar provides, rather than whether it fits into a specific win-win framework.

From this perspective, it is clear that more work is required to develop a holistic understanding of the ecological functions that biochar can contribute to the sustainable development (or ecological modernisation) of agriculture, both in China and globally. This could form part of the increasingly popular ecosystem services research agenda, which studies the processes through which the environment provides resources that are utilised by humans. These services tend to be split onto four categories, defined by the Millenium Ecosystem Assessment (2005) as:

- Supporting services: ecosystem services that are necessary for the production of all other ecosystem services
- Provisioning services: products obtained from ecosystems
- Regulating services: benefits obtained from the regulation of ecosystem processes
- Cultural services: nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences

Theoretically, biochar could contribute to supporting services (such as soil formation and nutrient cycling), provisioning services (such as food production), and regulating services (such as soil erosion regulation and climate regulation). Physical research on each of these individual services is increasingly available (Lehmann and Joseph, 2015), however more work is needed to combine and value these individual components within a systemic understanding of biochar's ecosystem service potential. Moreover, there are likley to be nuanced trade-offs that arise between the ecosystem services that biochar provides. For example, evidence suggests that biochar may have a short-term priming effect on soil carbon after it is applied, but over the longer term biochar application may suppress GHG emissions and sequester carbon in soil (Spokas and Reicosky, 2009; Van Zwieten *et al.*, 2010; Cross and Sohi, 2011; Zimmerman *et al.*, 2011). Additionaly, work from China suggests that biochar has the greatest yield impacts on dry-land crops such as maize, but has the greatest climate mitigation impacts on wet-land crops such as rice (Zhang *et al.*, 2012a,b), implying trade-offs between the "wins" that biochar provides when it is applied under different circumstances. This depth of analysis and acknowledgement of inherent trade-offs clearly contrasts with the more simplistic win-win approach, but is arguably an essential process through which to develop a nuanced understanding of what biochar offers to the field of sustainable development.

Although this broadening of scope to consider the wider ecosystem service contributions of biochar is a necessary research step, it remains predominantly grounded within a neoclassical and welfare economics approach to valuing and optimising environmental resources and services. However, recent research by Michael Grubb and colleagues (Grubb, 2014) suggests that whilst neoclassical and welfare economic approaches to environmental management are an essential viewpoint to consider, there are also two other theoretical approaches that can and must be integrated alongside this maintstream approach, in order to reach a truly sustainable development path. This was briefly touched upon in Chapter

2 (Theory and Methodology) and this final section returns to this idea, asking how future biochar research might learn from and apply the principles of this three-domain framework.

6.4 Three domains: the future for sustainable development?

In what is likely to become a seminal piece of work on the necessary pathways to sustainable development, Grubb (2014) outlines three domains that must be equally valued, understood, and applied if humanity is to stand a chance of averting serious climate change. These domains are called satisficing, optimising and transforming, and each map on to a theory of economics (behavioural economics; neoclassical/welfare economics; and institutional/evolutionary economics, respectively), a pillar of policy action (standards & engagement; markets & prices; and strategic investment, respectively) and an area on which each domain is intended to deliver (smarter choices; cleaner products & processes; innovation & infrastructure, respectively). Figure 6.1 provides a schematic of these domains and their respective policy pillars and deliverable areas [1].

To date, Grubb suggests, the "optimise" domain and its associated neoclassical & welfare theories of economics have dominated the western approach to environmental management and sustainable development. Whilst this domain is acknowledged as an essential part of the systemic response to the threats of climate change, it also cannot solve the problem alone. Indeed, Grubb (2014) states

[1] Grubb (2014), pg. 69

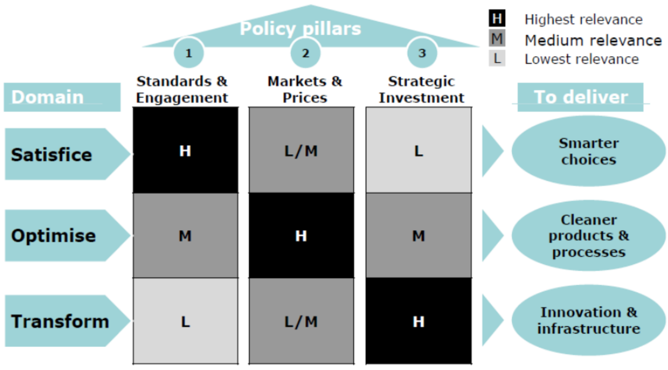


Figure 6.1: The three domains & their respective policy pillars and deliverable areas

that,

"...while traditional economics offers tools that may be adequate in considering problems of limited scope ("marginal" changes to the existing system) over bounded time periods and within individual countries, that toolbox is far too limited with respect to problems on the scale and timescale of energy and climate challenges." (Pg. 59)

On this basis, it seems that tackling "super wicked problems" (Rittel and Webber, 1973) such as climate change will require a broad, intergrated and multi-disciplinary approach, and it therefore makes sense to view the contribution of individual technologies, such as biochar, within this framework. For example, domain one ("satisficing") focuses on improving understanding of how human decision-making diverges from the neoclassical ideal of "rational economic man" using behavioural economics approaches. Related to biochar research, this could consider the psychology of farmer decisions to adopt biochar as an agricultural technology, and/or what strategies might be used to encourage biochar adoption where a holistic view of its ecosystem services suggests that this will have positive environmental and societal impacts. For example, should biochar be deployed in the most efficient manner possible, such as combined with existing inorganic fertiliser products, in order to reduce the decision-making and labour burdens on farmers? Or should it be marketed as an aspirational product for farmers in China who are looking to consolidate smaller land parcels into larger commercial plots, and are looking for something to give them a competitive edge in the produce markets?

Domain two ("optimising") considers the optimisation of existing systems, aiming to internalise environmental externalities using market instruments with a view to creating cleaner processes and products. Much of the discussion around biochar to date has been in this domain, including the valuation of its carbon sequestration services, and its ability to offset carbon-intensive inorganic fertiliser products. However, as discussed in Section 6.3, the indicators upon which biochar has so far been valued have been relatively narrow, and therefore a more holistic ecosystem services approach is warranted. Thus, although Grubb acknowledges that win-wins are one aspect of domain two, there are other approaches to optimising that exist and should be used within this domain.

Finally, domain three ("transforming"), focuses on evolutionary/institutional economic methods, and aims to understand the ways in which technological and institutional innovations are entwined and impact on the evolution of global systems. This domain looks beyond the marginal changes that typify domains one and two, expanding to consider broad-scale future scenarios that manage the systemic risks of climate change in a precautionary and integrated manner. Although biochar is a relatively small consideration for global-level climate change scenarios, future biochar research could contribute to some global scenario development exercises, ensuring that biochar's relative contributions are well understood in comparison to other technologies that may also have a place within the scenarios.

Figure 6.2 provides a schematic description of the suggested progression for biochar research from a relatively narrow win-win approach, through to the broader lense of ecosystem services within domain two, through to the larger systemic framework and strategy considerations of domain three. Notably, there are links maintained between the three domains in the final stage of this figure, indicating their interconnected nature and the ability for concepts to be applied across all three domains simultaneously. For example, although ecosystem services is most often applied through a domain two lense of marginal change and optimising ecosystem services for human welfare, it could equally be operationalised through a behavioural-change lense (domain one), or incorporated into global strategies of risk (domain three) related to, for example, ecosystem tipping points (Convention on Biological Diversity, 2010).

Importantly, although Figure 6.2 indicates the extent to which research on biochar must expand, it also places the win-win framework used in this thesis firmly within domain two, reinforcing the notion that neoclassical and welfare-economic analyses such as those included in this thesis are essential, though not sufficient, to further our understanding of how and where improved environmental management and sustainable development might be achieved. Indeed, in late 2011 when this PhD began, there was very little evidence available for the win-win potential of biochar from this perspective, and therefore this thesis has achieved its aims to contribute socio-economic research in this area.

This section has outlined the future research needs for biochar, and has highlighted the need for a much broader, inter-disciplinary research agenda going forward.

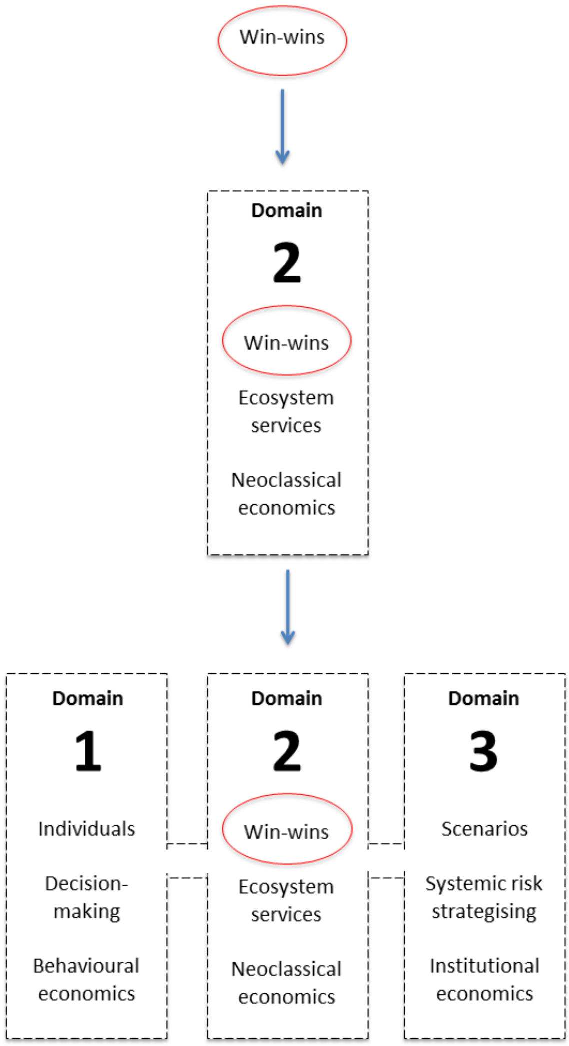


Figure 6.2: A schematic suggestion for biochar research progression

However, it is also pertinent to discuss potential conclusions for policy making that could be drawn from the work in this thesis. Therefore the following section discusses the potential to draw policy conclusions for the development of biochar products in China.

6.5 Implications for biochar policy

At present, from the results reported in this thesis, it could easily be argued that biochar has no place within China's agricultural policy-making framework. This is due both to the relatively few economic and environmental benefits that this research has reported for biochar as a competing use of China's straw, but also due to the current structure of China's farming system.

Focusing on this latter point, the adoption of new agricultural technologies by many of China's farmers is likely to be hindered by the pervasive influence of the "hukou" system, and its impact on societal structures across the country. The hukou is China's internal passport, which designates from birth whether an individual is born in a rural location or an urban location, and assigns them differing rights based on this status. Those with a rural hukou have rights to a small plot of land and the related agricultural subsidies, whereas those with an urban hukou are permitted to live and work in cities, accessing a variety of services ranging from being able to apply for a bus pass, all the way to healthcare access and the ability to enrol children in school (Wing Chan and Buckingham, 2008). These services are not available to those without an urban hukou, and this makes it very challenging to move a family from the rural to urban areas without hukou change. Those with a rural hukou can apply for an urban hukou, but this can be a difficult and costly process depending on which city a rural resident may wish to live in. The result is that working age rural adults tend to migrate off-farm to the urban areas, and work illegally in the cities, e.g., as unofficial taxi drivers. They send remittances back to the rural areas, thus increasing the incomes of rural families, making them less dependent on farming, but rarely enabling them to move off-farm.

This has important implications for agricultural development and the individual motivations of farmers to consider adopting new technologies for their farms. Firstly, it means that rural families do not move off-farm to the cities, thus maintaining their relatively small plots of land, and preventing the agglomeration of land into larger parcels that can be farmed commercially (Zhu, 2007). As a result, the majority of China's land remains managed by over 200 million smallholder farmers, averaging a land size of less than 1ha per household (Huang and Rozelle, 2015). Secondly, as rural families receive remittances from relatives working in urban areas, or find off-farm work in the local area, their income increases, their time on-farm decreases, and their reliance on the farm as a key livelihood activity decreases. As such, there is a decreasing motivation to rationalise farming inputs such as inorganic fertiliser, and there is also less time available to recycle organic wastes for cooking and/or animal feed. This has almost certianly contributed to China's widespread overuse of inorganic fertiliser and proliferation of post-harvest straw-burning in recent years.

As such, any policy recommendations for biochar must be made with careful consideration of this unique situation. In short, a vast number of smallholder farmers control huge swathes of China's land, with limited motivation to improve either the productivity or environmental impact of their farming practices.

With this in mind, it could easily be argued that biochar policy would be largely impotent in the face of such systemic challenges to sustainability. However, the hukou system is slowly changing (The Economist, 2015), and over time this may steadily allow greater land consolidation and an increase in commercial farmers managing large tracts of land, with improved motivation to invest in long-term soil quality and modern technologies. In addition, the urgency with which China needs to address environmental issues such as soil fertility and fertiliser overuse may increase to a point where there is a stronger need for biochar deployment and adoption than currently exists. Under these circumstances, biochar and associated policies may have a role to play, and therefore the remainder of this section discusses the types of policy recommendations that may be appropriate in these eventualities.

6.5.1 Suggestions for biochar policy in China

In considering what sort of policies might be appropriate, it is helpful to return to Grubb's three policy pillars: Standards & Engagement, Markets & Prices, and Strategic Investment (See Figure 6.1). The most relevant to biochar at the present time is the Strategic Investment pillar. The development of a biochar industry will need to be coordinated with the inorganic fertiliser industry, creating BMCC products that can simultaneously decrease inorganic fertiliser use, steadily improve soil organic carbon levels, and produce long-term soil fertility improvements across the country. Given the strong links between the Chinese government and the fertiliser production industry, strategic policies will be needed to ensure that fertiliser producers engage with research and outreach activities that maximise the successful creation and dissemination of biochar products.

As these strategic policies are developed and implemented, the Markets & Prices policy pillar will need to be carefully monitored. Developing BMCCs may be more expensive than standard inorganic fertiliser products, both due to the increased research needs compared to inorganic fertilisers, and also due to the added costs related to biomass sourcing, transport and pyrolysis. There is also likely to be higher risk and uncertainty in the development of BMCCs compared to the status quo. If the Chinese government is backing the wide-spread deployment of biochar, it will need to carefully create market and pricing incentives both for businesses to produce the BMCCs but also for the farmers to purchase and apply them. This may require subsidies, training programmes, and/or supporting agricultural standards and enforceable legislation about nutrient content of fertilisers and/or organic amendment standards.

Finally, the Standards & Engagement policy pillar should be used to ensure that any BMCC products meet rigorous quality assurance requirements, thus developing and maintaining consumer confidence in this new product. These product standards could also be accompanied by enforceable agricultural nutrient application standards, and food standards that indicate whether the food being bought has been produced on a farm where soil quality and fertility maintenance is assured.

Looking ahead, it could be wise for the Chinese government to begin approaching biochar development through Grubb's Strategic Investment policy pillar in the next few years, thus setting in motion a range of investment strategies and environmental goals that may provide the supporting back-drop to wide-spread adoption of biochar products in the future.

6.6 Conclusions and future research needs

This thesis set out to investigate whether biochar is an economic and environmental win-win solution for China's straw resources, from the perspective of ecological modernisation. The findings indicate that, within the current economic and policy climate, biochar is unlikely to be perceived as an attractive financial "win" option for farmers or businesses in China. Nor is it likely to be the most cost-effective climate change mitigation solution for governments to meet national carbon reduction targets, or for global institutions interested in funding climate change mitigation projects. Moreover, although some bioenergy win-win options have been identified from the research process, this final chapter has moved on to critique these findings within the broader consideration of what indicators can and should be included in the framing of a win-win solution.

Overall this thesis concludes that biochar cannot live up to the win-win expectations that it was initially framed within, however this does not mean that biochar has nothing to contribute to sustainable development or the future of Chinese agriculture. In fact, this thesis has highlighted the narrow nature of the win-win framework, and recommends that future biochar research expand beyond these limits to include assessments of a range of ecosystem services, alongside investigations into the behavioural aspects of biochar adoption by farmers, and the potential place of biochar within larger systemic frameworks of risk reduction around food production, climate change and soil fertility. Moreover, the preceding section further suggests that the Chinese government begin engaging with biochar research and development on a long-term, strategic basis. This ensures that, should conditions become more favourable for biochar adoption in future, China will be able to act in a timely and informed manner to ensure that biochar is integrated within agricultural systems in a way that both benefits farmers and the environment.

Whilst some win-win solutions are likely to exist, care must be taken to ensure that they are defined as win-win on the basis of their well understood and carefully considered intrinsic characteristics, rather than for their charismatic potential to impress funding bodies or gloss over difficult trade-offs. For biochar and bioenergy research alike, the future must lie in more holistic, methodologically diverse studies of these technologies and approaches to a variety of actors and to the global environmental system as a whole.

Chapter 7

Appendix 1: Sample questionnaire

Household survey questionnaire

Village: ………………… Interviewee (mother, father etc):………………… Age:……………….

1. What do you think about farming – is it important to your family?
Not at all Not very Somewhat Quite Very
Explain: ……..

Land

2. How is your land organised between allocated land, renting in and out, and contract farming?

	TOTAL land farmed	Allocated Land	Rent In	Rent Out	Contract
mu					
RMB					

3. Have you ever had your land allocation changed? If so when, and how much land……………………….
4. Would you like more land to grow more crops?..

Crops and Labour

5. How many mu do you plant of each crop?

	Crops:						
Land planted	mu						

6. What is your crop rotation? (Fill in on last page)
7. Have you changed this in the past five years?...
8. Are you or have you every been a member of a farmer cooperative or organisation?......................
If yes, what are the pros and cons to membership? ………………………………………………………………

9. What inputs do you use on your farm?

Fert ↓ Crop:							

10. In the case of trying new fertilisers would you say you:
Never try new ones Rarely try new ones Not sure Sometimes Try Always try

11. When was the last time you changed your fertiliser brand?
………

12. Did this change improve production? If yes, how much did the change cost, and how much did your yield increase? (i.e., what was the increase in profit from this change (y/mu)
………

13. Do you keep animals?

Animal	Number

14. Do you use manure from these animals?................

15. How does manure help the crops to grow?........................

....

16. Do you use machines/animals for farm tasks?

Machine/Animal	Task	Cost / Own?

Productivity

17. How productive do you think your farm is?

0 (no production) 1 2 3 4 5 6 7 8 9 10 (max)

<u>If they answer below 10,</u> are you satisfied with this level of productivity? ....

<u>If yes</u>, why don't they want more yields...

<u>If no</u>, how do you think you can improve production?...

Are you willing to change fertiliser to improve production?....................................

Are you willing to work harder to improve production? No A little bit Lots

18. Are there any problems that stop crops growing? (Rank for importance)

Crop				
Biggest problem				
2nd Biggest				
3rd Biggest				

19. When did you last have a bad year for yields? What was the yield vs. what was supposed to be?
....

20. How did you get through/recover from this *(e.g. relied on savings, helped by family, took out a loan etc.)*? ....

21. Is it possible to borrow money from friends or banks? Have you ever done this?
....

22. Have you ever used some kind of crop insurance to reduce risks for a bad year?..........................

Biomass and Wood

23. How much straw does each crop produce and what is it used for?

Crop	Use 1	% 1	Use 2	% 2	Use 3	% 3

24. Does anyone around here buy or collect biomass from farmers? What for? ....

25. Would you sell your biomass?...............
If yes, how much would you sell biomass for if you collect it?...........................(y/mu)
How much would you sell biomass for if someone else collects it…………(y/mu)

26. What is your main source of energy:

Fuel	Cost (y/month)	What it is used for
Gas		
Electricity		
Wood		
Other		

27. Do you grow trees / have access to wood?

Wood Type	Public or private?	Use 1 (%)	Use 2 (%)	Use 3 (%)

Crops
28. How much rice/maize/other crops do you eat / sell / buy each year?

Crop	*Last Yield (kg/RMB per mu)*	*Sale Price (per kg)*	*Profit made last harvest*

29. Where do you sell your crops? If far away, what is the means and cost of transport?
……

30. Have you ever had difficulty selling your crops?..

Demographics
32. Who lives in your household (eat from the same pot)?

Person	Age	Education	Job	Income RMB/year

33. Do you have any children/partners working away from home?

Person	Age	Education	Job	Income RMB/year	Send money home?

Chapter 8

Appendix 2: Supporting information to Chapter 4

This SI document includes text, tables and figures that provide further detail on the parameter values used to construct the CBA and LCA presented in this paper. The first half of the supplementary information refers to the CBA, and the second half refers to the LCA.

Technology Readiness Levels

S1 Technology Readiness Levels

Cost-Benefit Analysis SI

S2 Project Lifetime

S3 Project Finance

S4 Straw Collection Cost

S5 Capital and Operation Costs

Life-Cycle Analysis SI

Sensitivity Analysis SI

S19 Parameter values for CBA sensitivity analysis

S20 Parameter values for LCA sensitivity analysis

8.1 Technology Readiness Levels

S1 Descriptions of Technology Readiness Levels

TRL 1	Basic principles observed and reported
TRL 2	Technology concept and/or application formulated
TRL 3	Analytical and experimental critical function / characteristic proof-of-concept
TRL 4	Technology basic validation in a laboratory environment
TRL 5	Technology basic validation in a relevant environment
TRL 6	Technology model or prototype demonstration in a relevant environment
TRL 7	Technology prototype demonstration in an operational environment
TRL 8	Actual technology completed and qualified through test and demonstration
TRL 9	Actual technology qualified through successful mission operations

Table adapted from Mankins (1995) and UK Ministry of Defence (2014).

8.2 Cost-Benefit Analysis SI

S2 Project Lifetime

The briquetting, gasification and pyrolysis scenarios are compared for their economic viability by contrasting their project level net present value (NPV). Project lifetime is assumed to be 20 years, in accordance with Chinese bioenergy project timelines (Zhang *et al.*, 2014).

S3 Project Finance

Loans are assumed to be taken out for 75% of the project cost (Zhang *et al.*, 2014), and the opportunity cost of the remaining 25% capital investment is accounted for. Loan repayments are spread over 20 years. The discount and interest rates are set according to data on the Chinese economy (Federal Reserve Bank of St Louis, 2014; Trading Economics, 2014). Tax rates and tax breaks provided for bioelectricity projects are based on data from Chinese national policy documents (Zhang *et al.*, 2014)).

S4 Straw Collection Cost

Sanli New Energy Factory provides local straw collection agents with free access to baling machines, to enable straw collection. These agents then use their own tractors and trailers to transport the bailed straw to the straw collection depots. Thus, for Sanli the cost of straw collection is in the purchase and maintenance of balers, and in the payment to agents for the straw they deliver. Straw price data was combined from interviews at Sanli, in surrounding Henan villages, and from published academic data on straw pries for bioenergy projects (Zhang *et al.*, 2014). The average value across these data sources was \$45 Mg^{-1}. However due to the volatility of straw prices, a +/- 20% range was also included in the sensitivity analyses. The local government subsidy for straw collection (\$28 Mg^{-1}) was included on the basis of interview data from Henan. However it is not clear that this subsidy is widely available across all parts of China, and is likely to depend strongly on local government incentives for curtailing in-field straw burning.

S5 Capital and Operational Costs

A variety of data sources were used to price the capital costs of briquette machines, the gasification unit, the pyrolysis unit and the buildings to house them (Bridgwater *et al.*, 2002; Badger *et al.*, 2011; Zhang *et al.*, 2014). In each case where market data was collected online, three or more quotes were obtained for similar products, and then averaged to derive a mean value. Maintenance of these capital units, as with all machinery in all scenarios, is estimated at 5% of the capital value. Salvage value at the end of the lifetime is also fixed at 5%. Decommissioning costs were estimated at 2%.

Electricity requirements for the balers are derived from the supplier website (9.9 kWh odt^{-1}), and the cost of purchasing electricity from Chinas central grid is estimated as \$0.02 kWh^{-1}, as obtained from interviews at Sanli and around Henan villages. Start-up fuel requirements for gasification and pyrolysis are taken from Roberts *et al.* (2010) and costed according to fuel cost data from the World Bank (2014).

Staffing requirements for all scenarios also estimated from Sanli interviews, and priced according to labour market information (Clare *et al.*, 2014).
The CBA is calculated from the perspective of a potential investor, therefore the costs to a farmer of straw reincorporation and/or straw burning are not considered as part of the economic analysis.

S6 Pricing Outputs

Briquettes were valued according to their energy density (16MJ kg^{-1}; Roberts *et al.* (2010)) and the market value of an equivalent energy delivery from coal (Zhao and Yan, 2012; Bloomberg, 2014), which is the fossil fuel material that they would be offsetting.

Electricity outputs were valued using data on the base price for electricity from coal-fired power plants and the data on the subsidised prices provided for bioelectricity generated from agricultural wastes (Zhang *et al.*, 2014).

Biochar outputs were valued according to the latest meta-analytical data on biochars agronomic impact for a given application rate (Crane-Droesch *et al.*, 2013) and interview data on the agricultural market prices and cropping systems in Henan (Clare *et al.*, 2014).

8.3 Life Cycle Analysis SI

S7 LCA Functional Unit of Analysis

The functional unit of analysis for each LCA is one oven dry tonne (odt) of straw, assumed to be 40% maize and 60% wheat, to reflect the reported proportions of straw that Sanli New Energy Factory uses annually in its operations, and the cropping system of Henan province more generally.

S8 Global Warming Period (GWP)

The LCA considers the global warming potential of each scenario on a 100 year basis, according to calculated emissions of CO_2, CH_4 and N_2O. Carbon monoxide (CO) emissions were treated as equivalent to CO_2, as CO quickly oxidises to CO_2 once released to the atmosphere (Woolf *et al.*, 2010a). Due to uncertainties of global warming potentials for NO_x, SO_x, and particulate emissions, we are unable to include them in the 100year GWP estimate.

S9 Straw Cultivation and Removal

Emissions from agricultural inputs and activities involved in generating the grain-crops from which the straw is taken are not included in the LCA scenarios. This is because straw is a waste product that is generated as a by-product of food cultivation, which would exist in the absence of the S_{Briq}, S_{Gas}, or S_{Pyr} scenarios. At present, farmers do not receive payment for their straw, because it is seen as a nuisance by-product of little value. However it is worth noting that demand for biomass can change rapidly, therefore sensitivity analysis is used to model the effects of price changes (i.e., as a result of increased demand for a straw).

S10 Crop Residue Collection

One tonne of straw containing 40% maize and 60% wheat is assumed to contain 433160 grams of carbon (Li *et al.*, 2007). All subsequent calculations of emissions per tonne of feedstock are normalised, assuming this starting carbon content of 433160 grams per tonne. Carbon and energy contents of maize and wheat straw are taken from McKendry (2002b); Li *et al.* (2007); Roberts *et al.* (2010). Straw availability is calculated according to surveyed grain yields (Clare *et al.*, 2014) and grain:straw conversion ratios (Jiang *et al.*, 2012), accounting for 40% of wheat straw and 50% of maize straw to be left as crop cover (Scarlat *et al.*, 2010) in order to avoid negative impacts on soil organic carbon (SOC). The straw is collected and baled mechanically, and transported using a tractor and trailer (fuel requirements calculated using data from Dalgaard *et al.* (2001)). Distance for straw transportation from field to plant is calculated according to a methodology developed by French (1960) and subsequently modified by Roberts *et al.* (2010).

Embedded emissions of straw balers are included in this analysis, as Sanli purchases these to loan to straw collection agents. However embedded emissions within tractors and trailers are excluded, as these are already owned by the straw collection agents. It is assumed that 50 balers are required for the twenty year lifetime of the project, based on estimates of maximum machine lifetime, measured in hours of operation (Edwards, 2009). An average baler weight is calculated from Chinese agricultural equipment sales websites, and assumed to be predominantly constructed from steel. All steel accounted for in these calculations is assumed to be 70% recycled and 30% virgin (Wang, 2008; Roberts *et al.*, 2010).

Emissions embedded within storage buildings are also included. Building size is calculated according to the expected peak volume of straw that will be stored at any one time during the year (c.17,500 tonnes). Building construction materials are assumed to be concrete and steel. Finally, embedded emissions for four straw chopping machines are included. Machine weight is taken from a Chinese equipment website and assumed to be predominantly constructed from steel.
Emissions factors for virgin, recycled and stainless steel; concrete; virgin and recycled cast aluminium; and cast iron are taken from the GREET 1.8b spreadsheet (Wang, 2008).

S11 Straw Reincorporation (S_{Rein})

We assume that all carbon contained within the feedstock is released as CO_2 over the 100 year time period (Knoblauch *et al.*, 2011). In agricultural soils, where straw is reincorporated and then turned twice a year for planting, it is extremely unlikely that any carbon in this feedstock would remain stabilised in soil after 100 years ((Lehmann *et al.*, 2006).

S12 Straw Burning (S_{Burn})

Data on CO_2, CH_4 and N_2O emissions from open field straw burning is taken from Li *et al.* (2007). Monte Carlo methods were used to generate distributions using mean and standard deviation emissions values for maize and wheat straw. These distributions were plotted, normalised, and combined in a 4:6 ratio according to the reported maize:wheat straw processed at Sanli.

S13 Briquetting (S_{Briq})

The briquette machines process 2.75 tonnes of straw per hour, using 9.9kWh odt^{-1} feedstock. Electricity emissions are estimated according to a grid emissions factor for the north regional grid, which includes Henan within its service area (World Resources Institute, 2014). It is assumed that the 20 year project lifetime will require 100 briquette machines, each processing 5,600 tonnes of straw during its working lifetime. Each machine weighs 1.6 tonnes and is made primarily from steel. The briquette machines also require a shelter, which is estimated to require 1 tonne of steel and 2.5 tonnes of concrete.

Mean and standard deviation values for emissions of CO_2 and CH_4 from burning straw briquettes and coal in domestic stoves is taken from Zhang *et al.* (2000) and Wang *et al.* (2013a). Monte Carlo method was used to generate distributions according to these reported mean and standard deviation values. These distributions were plotted, normalised, and combined in a 4:6 ratio according to the reported maize:wheat straw ratio. The coal emissions distribution was then combined with an estimate for emissions required to source coal (Hill *et al.*, 2013), which accounted for 15% of the total coal emissions. In the absence of available data on emissions related to sourcing coal in China, the coal sourcing emissions estimate is based on UK data.

S14 Equipment, Fuel and Buildings

For S_{Gas} and S_{Pyr}, the emissions associated with equipment (i.e., construction of gasification and pyrolysis units), start-up fuel, and buildings are assumed to be the same. This is because each unit processes the same quantity of straw, and there is little data available to justify distinctions in estimates in construction materials needed for each unit. Data from Mann and Spath (1997) is used to calculate the volumes of materials needed, and we find that our materials estimates are similar to those reported by Roberts *et al.* (2010) (See Table 8.1).

Emissions from plant decommissioning and the transport and recycling of materials have previously been found to be negligible (Lombardi, 2003) and are therefore not included in the analysis.

Table 8.1: Comparison of tonnes of materials estimates with Roberts et al. (2010)

Material	Our Estimate (tonnes per pyrolysis unit)	Roberts et al. estimate
Concrete	1748.2	1759.5
Steel	653.9	558.5
Aluminium	5.1	7.3
Iron	7.6	3.7

Estimated start-up fuel for each scenario is based on Saft (2007), with associated emissions for natural gas calculated from the a report by the Biomass Energy Centre (2014). Finally we assume that four tonnes of steel and ten tonnes of concrete are used to build a shelter that houses the gasification/pyrolysis unit. Again, emissions associated with each material are based on estimates from the GREET 1.8b spreadsheet (Wang, 2008).

S15 Direct Emissions

Guidance for estimating emissions from gasification and pyrolysis of straw is taken from Lu and Zhang (2010a), who report emissions from the combustion of syngas created from biomass gasification. The latter was used as an estimate of direct emissions for syngas generated from both gasification and pyrolysis, in the absence of data specific to the combustion of pyrolysis-derived syngas. It should be noted that other pyrolysis LCA papers have assumed complete combustion of syngas to carbon dioxide and water (Roberts *et al.*, 2010; Hammond *et al.*, 2011).

Lu and Zhang (2010a) provide mean values, with no measure of variance. Therefore variance was estimated by calculating the percentage of initial feedstock carbon released to atmosphere during combustion of syngas derived from biomass (ca.80%) and then using the reported ratio of CO_2:CH_4 (assuming that this is a guideline to the efficiency and/or cleanliness of the burn process) within a Monte Carlo generated distribution whose variance was determined by varying this ratio by 50% in each direction.

S16 Energy Offsets

In order to calculate energy offsets for each LCA scenario, the efficiency of each system must be calculated. These calculations are based on a variety of high quality published articles, government agency reports and contact with commercial companies (Quaak *et al.*, 1999; Bridgwater *et al.*, 2002; EPA, 2008; Lu and Zhang, 2010a; Roberts *et al.*, 2010; Clarke Energy, 2014; Weifang Naipute Gas Genset Co. Ltd., 2014).

Figures S1 and S2 provide Sankey diagrams for the conversion of straw to cleaned syngas for gasification and pyrolysis systems, respectively. The cleaned syngas is then converted to electricity via combustion in a gas engine, at 32.5% efficiency (Clarke Energy, 2014; Weifang Naipute Gas Genset Co. Ltd., 2014).

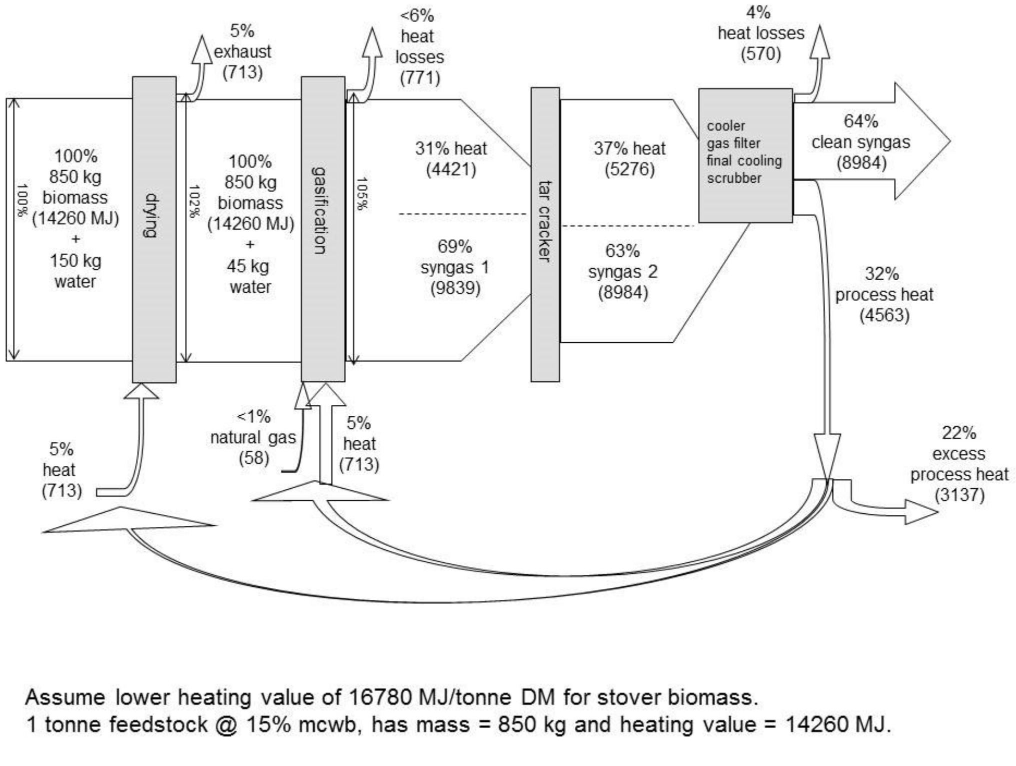
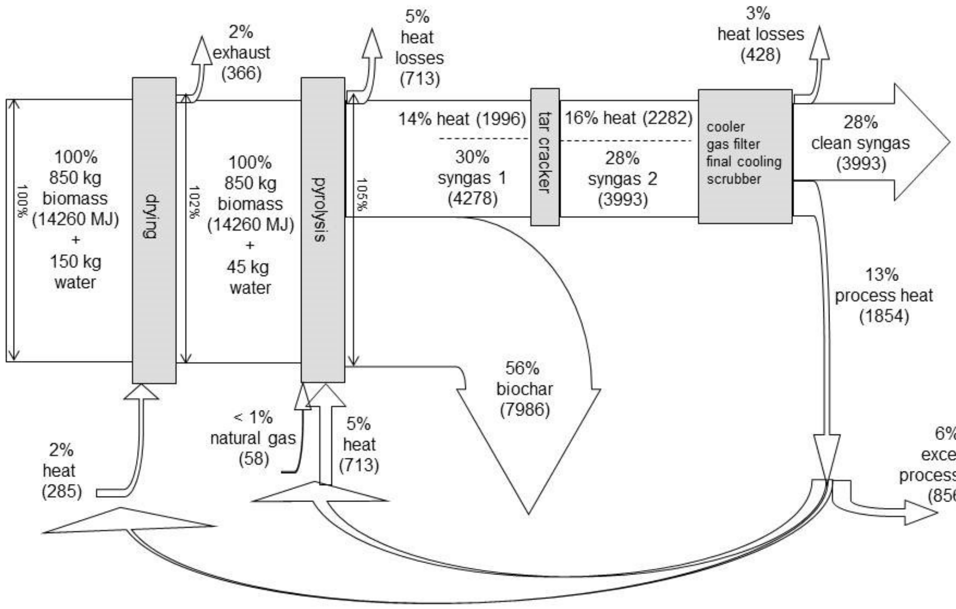


Figure 8.1: Sankey diagram of the gasification conversion of straw to syngas



Assume lower heating value of 16780 MJ/tonne DM for stover biomass.
1 tonne feedstock @ 15% mcwb, has mass = 850 kg and heating value = 14260 MJ.

Figure 8.2: Sankey diagram of the slow pyrolytic conversion of straw to syngas and biochar

Table 8.2 outlines the median and quartile ranges of estimates for MWh produced per tonne of feedstock, using the efficiency estimates that are displayed in the Sankey diagrams above.

Table 8.2: Median and 95% confidence intervals for MWh produced per tonne of feedstock processed for gasification and pyrolysis units

	Gasification (MWh odt^{-1})	Pyrolysis (MWh odt^{-1})
Median	0.95	0.38
95% confidence interval	(0.73, 1.18)	(0.31, 0.47)

Offsets are then calculated by estimating the equivalent emissions that would be generated had the same number of MWh been provided by the Chinese northern grid, with an estimated emissions factor of 1.13Mg CO_2e MWh^{-1} (World Resources Institute, 2014).

S17 Carbon sequestration

In S_{Gas} it is assumed that 80% of feedstock carbon is emitted as CO_2 or CH_4 during gasification (Lu and Zhang, 2010b), leaving 20% of feedstock carbon remaining in the ashy char by-product. This char is assumed to be 90% stable (Singh *et al.*, 2012). In S_{Pyr} it is assumed that 29.6% of feedstock weight is converted to biochar (Roberts *et al.*, 2010), which contains 71% carbon (Thomsen *et al.*, 2011), 80% of which remains in the soil after 100 years (Singh *et al.*, 2012). The higher stability of S_{Gas} char is based on the positive relationship between high temperatures with long-term carbon stability (Crombie *et al.*, 2013).

S18 Monte Carlo Analysis

Monte Carlo analysis was used to generate distributions and estimates of uncertainty around each parameter used in the LCA scenarios. Some data sources provided measures of variance around their point estimates, in which case these were used to generate distributions (as in the case of emissions of in-field straw burning, briquette burning, and various estimates of process efficiency for the combustion, gasification and pyrolysis processes).

Where no variance points were given, distributions were created around the mean point estimates provided, using a standard deviation based on a 95% confidence interval +/- 50%. In the absence of a pre-existing estimate of variance, it is appropriate to use a broad range of possibilities to test the sensitivity of the analysis to such parameter variability.

All distributions were then reported according to their median value, rather than the mean. This is because some distributions were skewed, making it more appropriate to use the median as comparative measure of location, rather than the mean.

8.4 Sensitivity Analysis Supporting Information

S19 Parameter values for CBA sensitivity analysis

		Baseline	Range
S_{Briq}	Straw price (\$ Mg^{-1})	49	39 - 59
	Local subsidy (\$ Mg^{-1} straw burn avoided)	28	22 - 34
	Capital cost (m\$ total)	0.32	0.26 0.38
	Labour cost (\$ $hour^{-1}$)	3	2.4 3.6
	Briquette sale price	57	46 - 67
S_{Gas}	Straw price (\$ Mg^{-1})	49	39 - 59
	Local subsidy (\$ Mg^{-1} straw burn avoided)	28	22 - 34
	Capital cost (m\$ total)	6.34	5.07 7.60
	Labour cost (\$ $hour^{-1}$)	3	2.4 3.6
	Bioelectricity price (\$ kWh^{-1} bioelectricity)	0.12	0.10 0.14
S_{Pyr}	Straw price (\$ Mg^{-1})	49	39 - 59
	Local subsidy (\$ Mg^{-1} straw burn avoided)	28	22 - 34
	Capital cost (m\$ total)	7.6	6.10 9,16
	Labour cost (\$ $hour^{-1}$)	3	2.4 3.6
	Bioelectricity price (\$ kWh^{-1} bioelectricity)	0.12	0.10 0.14
	Biochar price (\$ Mg^{-1})	110	88 - 132

S20 Parameter values for LCA sensitivity analysis

		Baseline	Range
S_{Briq}	Straw collection ($MgCO_2e$ odt^{-1} feedstock)	0.05	0.04 0.05
	Machinery ($MgCO_2e$ odt^{-1} feedstock)	0.01	0.01-0.01
	Briquette combustion ($MgCO_2e$ odt^{-1} feedstock)	1.67	1.33 2.0
	Coal briquette offsets ($MgCO_2e$ odt^{-1} feedstock)	-1.44	0.57
S_{Gas}	Straw collection ($MgCO_2e$ odt^{-1} feedstock)	0.05	0.04 0.05
	Machinery ($MgCO_2e$ odt^{-1} feedstock)	0.01	0.01-0.01
	Straw gasification ($MgCO_2e$ odt^{-1} feedstock)	1.28	1.2 1.53
	Electricity offsets ($MgCO_2e$ odt^{-1} feedstock)	-0.91	0.36
S_{Pyr}	Straw collection ($MgCO_2e$ odt^{-1} feedstock)	0.05	0.04 0.05
	Machinery ($MgCO_2e$ odt^{-1} feedstock)	0.01	0.01-0.01
	Straw pyrolysis ($MgCO_2e$ odt^{-1} feedstock)	0.66	0.53 0.79
	Electricity offsets ($MgCO_2e$ odt^{-1} feedstock)	-0.34	0.14
	Fertiliser avoidance offsets ($MgCO_2e$ odt^{-1} feedstock)	-0.39	0.16
	Biochar stability (% over 100 years)	80	64 - 96

Chapter 9

Appendix 3: Supporting information to Chapter 5

The following text provides a detailed description of the model used in Chapter 5 to calculate the annual bioenergy generation potential (TWh) from China's agricultural residues. The first three steps are completed using ArcGIS software, whilst steps four to seven are completed using R statistical analysis software.

- Assign straw residue data to map pixels. The agricultural residue data was assigned to geographical units using a farmland distribution map at 1:100,000 scale, obtained from the Resources and Environmental Sciences Data Centre (RESDC) and the Chinese Academy of Sciences. Straw energy values (MJ) were assigned to each geographic unit (1000m 1000m pixel) based on the area of farmland contained within each pixel, assuming that all rice is allocated to wet land, and that maize and wheat are allocated equally between dry and wet land:

$P_{i,j,t} = \frac{L_{i,j,t}}{\sum L_{i,j,t}} \times E_{j,t}$

where P = the energy (MJ) contained within pixel i, of county j, of land type t (wet vs. dry) , L = area of land contained within pixel i, of county j, of land type t, and E = the total energy (MJ) contained within county j, for land type t.

- Place powerstations on the map and construct 20km and 50km straw collection radii around each station

- Calculate straw energy (MJ) available in for each radius (20km and 50km) and under each straw removal scenario (S1-S3) for each powerstation. Export the resulting six straw MJ availability values to a dataset that can be analysed in R.

- Calculate technical TWh generation potential for each of the six radius-straw removal scenario combinations using data on powerstation installed capacity (GW), estimated energy conversion efficiency, annual operating time, straw MJ available within a given radius, and a variety of cofiring ratios.

- Calculate the relative internal rate of return (IRR) for each powerstation when generating electricity using straw biomass instead of coal, with and without the bioenergy feed-in-tariff. This is done by constructing a ten year cash-flow series accounting for the initial costs of retrofitting the powerstations to accept straw feedstocks, and then annual operating costs of sourcing coal/straw feedstocks, income from electricity generation, efficiency reductions from using straw instead of coal, and changes to fly ash sales.

- Identify which powerstations can cofire straw at IRRs of 8% or greater, and then sum the total annual TWh of bioenergy that would be produced from these powerstations at various cofiring rates.

- Vary selected financial parameters used for the IRR calculations to determine the sensitivity of the overall TWh generation result in the event of changes to these parameter values.

References

Al-Mansour, F. and Zuwala, J. (2010). An evaluation of biomass co-firing in Europe. *Biomass and Bioenergy*, **34**, 620–629.

Asai, H., Samson, B.K., Stephan, H.M., Songyikhangsuthor, K., Homma, K., Kiyono, Y., Inoue, Y., Shiraiwa, T. and Horie, T. (2009). Biochar amendment techniques for upland rice production in Northern Laos. *Field Crops Research*, **111**, 81–84.

Atkinson, C.J., Fitzgerald, J.D. and Hipps, N.A. (2010). Potential mechanisms for achieving agricultural benefits from biochar application to temperate soils: a review. *Plant and Soil*, **337**, 1–18.

Badger, P., Badger, S., Puettman, M., Steele, P. and Cooper, J. (2011). Techno-economic analysis: preliminary assessment of pyrolysis oil production costs and material energy balance associated with a transportable fast pyrolysis system. *bioResources*, **6**, 34–47.

Baker, S. (2007). Sustainable development as symbolic commitment: Declaratory politics and the seductive appeal of ecological modernisation in the European Union. *Environmental Politics*, **16**, 297–317.

Bapat, H., Manahan, S.E. and Larsen, D.W. (1999). An activated carbon product prepared from milo (Sorghum vulgare) grain for use in hazardous waste gasification by ChemChar cocurrent flow gasification. *Chemosphere*, **39**, 23–32.

Barney, G.O. (1980). *The Global 2000 Report to the President of the US: Entering the 21st Century: The Technical Report*. Elsevier.

Barrow, C. (2012). Biochar: potential for countering land degradation and for improving agriculture. *Applied Geography*, **34**, 21–28.

Basu, P., Butler, J. and Leon, M.A. (2011). Biomass co-firing options on the emission reduction and electricity generation costs in coal-fired power plants. *Renewable Energy*, **36**, 282–288.

Bender, D. and McCarl, B. (1992). *Energy crop selection using linear programming.*. Elsevier Science.

Berger, T. (2001). Agent-based spatial models applied to agriculture: a simulation tool for technology diffusion, resource use changes and policy analysis. *Agricultural Economics*, **25**, 245–260.

Bian, R., Chen, D., Liu, X., Cui, L., Li, L., Pan, G., Xie, D., Zheng, J., Zhang, X., Zheng, J. and Chang, A. (2013). Biochar soil amendment as a solution to prevent Cd-tainted rice from China: Results from a cross-site field experiment. *Ecological Engineering*, **58**, 378–383.

Biederman, L. and Harpole, W.S. (2013). Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. *Global Change Biology Bioenergy Bioenergy*, **5**, 202–214.

Biggs, S.D. (1990). Agricultural Source of Innovation and Technology Model of Promotion. *World Development*, **18**, 1481–1499.

Biomass Energy Centre (2014). Carbon Emissions of Different Fuels. `http://www.biomassenergycentre.org.uk/portal/page?\_pageid=75,163182\&\_dad=portal\&\_schema=PORTAL`, accessed: 2014-05-23.

Birkhaeuser, D., Evenson, R.E. and Feder, G. (1991). The economic impact of agricultural extension: A review. *Economic development and cultural change*, 607–650.

Black, D. (2010). Biochar: Running the numbers. `http://www.bbc.co.uk/news/science-environment-12008573`, accessed: 2012-01-02.

Bloomberg (2012). China Sets Waste-to-Power Price Double That of Coal-Fired Plants. `http://www.bloomberg.com/news/2012-04-10/china-sets-waste-to-power-price-double-that-of-coal-fired-plants.html`, accessed: 2014-07-01.

Bloomberg (2014). China Benchmark Spot Coal Price Drops First Time in Three Months. `http://www.bloomberg.com/news/2014-01-06/china-benchmark-spot-coal-price-drops-first-time-in-three-months.html`, accessed: 2014-10-03.

Blowers, A. (1997). Environmental policy: ecological modernisation or the risk society? *Urban studies*, **34**, 845–872.

Blühdorn, I. (2001). Reflexivity and self-referentiality: on the normative foundations of ecological communication. *Critical Studies*, **16**, 181–201.

Boyle, K.J. and Bishop, R.C. (1987). Valuing wildlife in benefit-cost analyses: A case study involving endangered species. *Water resources research*, **23**, 943–950.

Bridgwater, A. (2009). Technical and economic assessment of thermal processes for biofuels. *Life Cycle and Techno-Economic Assessment of the Northeast Biomass to Liquids Project. NNFCC project 8*.

Bridgwater, A., Toft, A. and Brammer, J. (2002). A techno-economic comparison of power production by biomass fast pyrolysis with gasification and combustion. *Renewable and Sustainable Energy Reviews*, **6**, 181–246.

Brown, R. (2009). Biochar Production Technology. In J. Lehmann and S. Joseph, eds., *Biochar for Environmental Management - Science and Technology*, 127–146, Earthscan, London, UK.

Brown, T.R., Wright, M.M. and Brown, R.C. (2010). Estimating profitability of two biochar production scenarios : slow pyrolysis vs fast pyrolysis. *Society of Chemical Industry*, **5**, 54–68.

Bruinsma, J. *et al.* (2003). *World agriculture: towards 2015/2030. An FAO perspective.*. Earthscan Publications Ltd.

Campbell, B.M. (2009). Beyond Copenhagen: REDD+, agriculture, adaptation strategies and poverty. *Global Environmental Change*, **19**, 397–399.

Cao, G., Zhang, X. and Zheng, F. (2006). Inventory of black carbon and organic carbon emissions from China. *Atmospheric Environment*, **40**, 6516–6527.

Cao, G., XiaoYe, Z., YaQiang, W. and FangCheng, Z. (2008). Estimation of emissions from field burning of crop straw in China. *Chinese Science Bulletin*, **53**, 784–790.

Carter, S. and Shackley, S. (2011). Biochar Stoves: an innovation studies perspective. *UK Biochar Research Centre, University of Edinburgh*.

Cayuela, M.L., Sánchez-Monedero, M.A., Roig, A., Hanley, K., Enders, A. and Lehmann, J. (2013). Biochar and denitrification in soils: when, how much and why does biochar reduce N2O emissions? *Scientific reports*, **3**, 1732.

Cernansky, R. (2015). Agriculture: State-of-the-art soil. *Nature News and Comment*, **517**, 258–260.

Chambers, R. and Conway, G. (1992). *Sustainable rural livelihoods: practical concepts for the 21st century*. Institute of Development Studies (UK).

Chan, K., Van Zwieten, L., Meszaros, I., Downie, A. and Joseph, S. (2008). Agronomic values of greenwaste biochar as a soil amendment. *Soil Research*, **45**, 629–634.

Chen, J. (2007). Rapid urbanization in China: A real challenge to soil protection and food security. *Catena*, **69**, 1–15.

Chen, W. and Xu, R. (2010). Clean coal technology development in China. *Energy Policy*, **38**, 2123–2130.

China Centre for Modernisation Research (2007). *China Modernization Report 2007: Study of Ecological Modernization*. Beijing: Beijing University Press.

China Electricity Council (2014). China Electric Power Yearbooks. `http://hvdc.chinapower.com.cn/membercenter/yearbookcenter/content.asp?user=yearbook\&columnid=122`, accessed: 2012-10-09.

Christoff, P. (1996). Ecological modernisation, ecological modernities. *Environmental Politics*, **5**, 476–500.

Clare, A., Barnes, A., McDonagh, J. and Shackley, S. (2014). From rhetoric to reality: farmer perspectives on the economic potential of biochar in China. *International Journal of Agricultural Sustainability*, **12**, 440–458.

Clarke Energy (2014). GE Jenbacher Type-6 Engine. `http://www.clarke-energy.com/gas-engines/type-6-gas-engines/;http://www.clarke-energy.com/gas-type/synthesis-gas-syngas/`, accessed: 2014-03-27.

Convention on Biological Diversity (2010). Global Biodiversity Outlook 3. In *Montréal, Canada: Secretariat of the Convention on Biological Diversity.(http://gbo3. cbd. int/) Phil. Trans. R. Soc. B*, vol. 9.

Cornelissen, G., Martinsen, V., Shitumbanuma, V., Alling, V., Breedveld, G.D., Rutherford, D.W., Sparrevik, M., Hale, S.E., Obia, A. and Mulder, J. (2013). Biochar effect on maize yield and soil characteristics in five conservation farming sites in zambia. *Agronomy*, **3**, 256–274.

Cornot-Gandolphe, S. (2014). China's Coal Market: Can Beijing tame 'King Coal'? Tech. Rep. December, The Oxford Insitute for Energy Studies, Oxford, UK.

Crane-Droesch, A., Abiven, S., Jeffery, S. and Torn, M.S. (2013). Heterogeneous global crop yield response to biochar: a meta-regression analysis. *Environmental Research Letters*, **8**, 44–49.

Crombie, K., Mašek, O., Sohi, S.P., Brownsort, P. and Cross, A. (2013). The effect of pyrolysis conditions on biochar stability as determined by three methods. *Global Change Biology Bioenergy*, **5**, 122–131.

Cross, A. and Sohi, S.P. (2011). The priming potential of biochar products in relation to labile carbon contents and soil organic matter status. *Soil Biology and Biochemistry*, **43**, 2127–2134.

Cross, A. and Sohi, S.P. (2013). A method for screening the relative long-term stability of biochar. *Global Change Biology Bioenergy Bioenergy*, **5**, 215–220.

Cuiping, L., Chuangzhi, W. and Haitao, H. (2004). Study on the distribution and quantity of biomass residues resource in China. *Biomass and Bioenergy*, **27**, 111–117.

Dalgaard, T., Halberg, N. and Porter, J.R. (2001). A model for fossil energy use in danish agriculture used to compare organic and conventional farming. *Agriculture, Ecosystems & Environment*, **87**, 51–65.

Daly, H.E. (1990). Toward some operational principles of sustainable development. *Ecological Economics*, **2**, 1–6.

de Brauw, A., Huang, J., Zhang, L. and Rozelle, S. (2012). The Feminization of Agriculture with Chinese Characteristics. *IFPRI Discussion Paper 01189*.

DECC (2008). The co-utilisation of waste biomass with coal: The application of co-firing in Chinese power boilers. Tech. rep., Department for Energy and Climate Change, London, UK.

Deininger, K. and Jin, S. (2005). The potential of land rental markets in the process of economic development: Evidence from China. *Journal of Development Economics*, **78**, 241–270.

Dent, J., Harrison, S. and Woodford, K. (1986). *Farm Planning with Linear Programming: Concept and Practice*. Butterworths.

Ding, C. (2008). Land policy reform in China : assessment and prospects. *Land Use Policy*, **20**, 109–120.

Dobson, A. (1998). *Justice and the Environment: Conceptions of Environmental Sustainability and Dimensions of Social Justice*. Oxford University Press., Oxford.

Dong, N. (2012). Support mechanisms for cofiring secondary fuels. Tech. rep., IEA Clean Coal Centre, London, UK.

Downie, A., Munroe, P., Cowie, A., Van Zwieten, L. and Lau, D.M.S. (2012). Biochar as a Geoengineering Climate Solution: Hazard Identification and Risk Management. *Critical Reviews in Environmental Science and Technology*, **42**, 225–250.

Dryzek, J. (2013). *The politics of the earth: environmental discourses (Third Ed.)*. Oxford University Press, Oxford.

Duan, F., Liu, X., Yu, T. and Cachier, H. (2004). Identification and estimate of biomass burning contribution to the urban aerosol organic carbon concentrations in Beijing. *Atmospheric Environment*, **38**, 1275–1282.

Edward-Jones, G. (2006). Modelling farmer decision-making: concepts, progress and challenges. *Animal Science*, **82**, 783–790.

Edwards, W. (2009). Estimating Farm Machinery Costs. `http://www.extension.iastate.edu/agdm/crops/html/a3-29.html`, accessed: 2014-01-07.

EIA (2015). International Energy Statistics. `http://www.eia.gov/cfapps/ipdbproject/iedindex3.cfm?tid=1\&pid=1\&aid=8\&cid=CH,\&syid=1980\&eyid=2012\&unit=MMTCD`, accessed: 2015-02-02.

EPA (2008). Direct Emissions from Stationary Combustion Sources. Tech. Rep. May, Climate Leaders GHG Inventory Protocol, EPA, USA.

Ernsting, A. and Smolker, R. (2009). Biochar for climate change mitigation: fact or fiction. Tech. rep., Biofuelwatch.

Fan, M., Shen, J., Yuan, L., Jiang, R., Chen, X., Davies, W.J. and Zhang, F. (2012). Improving crop productivity and resource use efficiency to ensure food security and environmental quality in China. *Journal of experimental botany*, **63**, 13–24.

Fan, S. and Chan-Kang, C. (2005). Is small beautiful? Farm size, productivity, and poverty in Asian agriculture. *Agricultural Economics*, **32**, 135–146.

Federal Reserve Bank of St Louis (2014). Discount rate for China. `http://research.stlouisfed.org/fred2/series/INTDSRCNM193N`, accessed: 2014-12-04.

Fei, L., Dong, S., Xue, L., Liang, Q. and Yang, W. (2011). Energy consumption-economic growth relationship and carbon dioxide emissions in China. *Energy Policy*, **39**, 568–574.

Field, J.L., Keske, C.M.H., Birch, G.L., DeFoort, M.W. and Cotrufo, M.F. (2013). Distributed biochar and bioenergy coproduction: a regionally specific case study of environmental benefits and economic impacts. *GCB Bioenergy*, **5**, 177–191.

Flyvbjerg, B. (2006). Five misunderstandings about case-study research. *Qualitative inquiry*, **12**, 219–245.

Freeman, A. (1982). *Air and water pollution control: a benefit-cost assessment*. John Wiley and Sons, New York, NY.

French, B. (1960). Some considerations in estimating assembly cost functions for agricultural processing operations. *Journal of Farm Economics*, **62**, 767–778.

Gale, F., Lohmar, B. and Tuan, F. (2005). China's New Farm Subsidies. *Economic Research Service/USDA*.

Galinato, S.P., Yoder, J.K. and Granatstein, D. (2011). The economic value of biochar in crop production and carbon sequestration. *Energy Policy*, **39**, 6344–6350.

Gan, L. and Yu, J. (2008). Bioenergy transition in rural China: Policy options and co-benefits. *Energy Policy*, **36**, 531–540.

Garnett, T., Appleby, M., Balmford, A., Bateman, I., Benton, T., Bloomer, P., Burlingame, B., Dawkins, M., Dolan, L., Fraser, D. *et al.* (2013). Sustainable intensification in agriculture: premises and policies. *Science*, **341**, 33–34.

Gibbons, J. and Ramsden, S. (2008). Integrated modelling of farm adaptation to climate change in east anglia, uk: scaling and farmer decision making. *Agriculture, Ecosystems & Environment*, **127**, 126–134.

Giorgi, L. and Redclift, M. (2000). European environmental research in the social sciences: research into ecological modernization as a boundary object. *European Environment*, **10**, 12–23.

Glaser, B., Haumaier, L., Guggenberger, G. and Zech, W. (2001). The 'Terra Preta' phenomenon: A model for sustainable agriculture in the humid tropics. *Naturwissenschaften*, **88**, 37–41.

Glaser, B., Lehmann, J. and Zech, W. (2002). Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal - a review. *Biology and Fertility of Soils*, **35**, 219–230.

Glaser, B., Parr, M., Braun, C. and Goodspeed, K. (2009). Biochar is carbon negative. *Nature Geoscience*, **2**, 210–219.

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M. and Toulmin, C. (2010). Food security: the challenge of feeding 9 billion people. *Science*, **327**, 812–818.

Goodall, C. (2010). *Ten Technologies to Save the Planet: Energy Options for a Low-Carbon Future*. Greystone Books Ltd.

Gosens, J. (2015). Biopower from direct firing of crop and forestry residues in China: A review of developments and investment outlook. *Biomass and Bioenergy*, **73**, 110–123.

Grieve, R.H. (2004). Appropriate Technology in a Globalizing World. *International Journal of Technology Management & Sustainable Development*, **3**, 173–187.

Grubb, M. (2014). Planetary economics. *Taylor Francis/Routledge, London, UK*.

Guo, M., Jia, X., Huang, J., Kumar, K.B. and Burger, N.E. (2015). Farmer field school and farmer knowledge acquisition in rice production: Experimental evaluation in China. *Agriculture, Ecosystems & Environment (DOI: 10.1016/j.agee.2015.02.011)*.

Haefele, S. (2007). Black Soil, green rice. *Rice Today*, **4**, 26–27.

Hajer, M. and Versteeg, W. (2005). A decade of discourse analysis of environmental politics: Achievements, challenges, perspectives. *Journal of Environmental Policy & Planning*, **7**, 175–184.

Hammond, J., Shackley, S., Sohi, S. and Brownsort, P. (2011). Prospective life cycle carbon abatement for pyrolysis biochar systems in the UK. *Energy Policy*, **39**, 2646–2655.

Han, J., Mol, A.P., Lu, Y. and Zhang, L. (2008). Small-scale bioenergy projects in rural China: Lessons to be learnt. *Energy Policy*, **36**, 2154–2162.

Hanley, N., Spash, C. and Cullen., R. (1993). *Cost-benefit analysis and the environment*. Edward Elgar, Aldershot, UK.

Hill, N., Venfield, H., Dun, C. and James, K. (2013). Government GHG Conversion Factors for Company Reporting : Methodology Paper for Emission Factors. Tech. Rep. July, DEFRA, London, UK.

Hu, Z. and Zhang, D. (2006). China: Country Pasture/Forage Resource Profiles. Tech. rep., Food and Agriculture Organisation (FAO).

Huang, J. and Rozelle, S. (2015). The Role of Agriculture in China's Development: Performance, Determinants of Successes and Future Challenges. In P. Shome and P. Sharma, eds., *Emerging Economies*, 67–88, Springer India.

Huang, J., Qiao, F. and Zhang, L. (2000). Farm Pesticide, Rice Production, and Human Health. Tech. rep., Economy and Environment Program for Southeast Asia (EEPSEA).

Huang, J., Hu, R., Cao, J., Rozelle, S. and Korea, S. (2008). Training programs and in-the-field guidance to reduce China's overuse of fertilizer without hurting profitability. *Journal of Soil and Water Conservation*, **63**, 163–165.

Huang, J., Gao, L. and Rozelle, S. (2012a). The effect of off-farm employment on the decisions of households to rent out and rent in cultivated land in China. *China Agricultural Economic Review*, **4**, 5–17.

Huang, J., Wang, X. and Qiu, H. (2012b). Small-scale farmers in China in the face of modernisation and globalisation. *Centre for Chinese Agricultural Policy, Report produced for IIED & Hivos*.

Huang, Y., McIlveen-Wright, D., Rezvani, S., Wang, Y., Hewitt, N. and Williams, B. (2006). Biomass co-firing in a pressurized fluidized bed combustion (PFBC) combined cycle power plant: A techno-environmental assessment based on computational simulations. *Fuel Processing Technology*, **87**, 927–934.

IEA (2012). Cofiring high ratios of biomass with coal. Tech. rep., IEA Clean Coal Centre, London, UK.

Industrial Efficiency Policy Database (2014). China: Energy Development Plan of the 12th Five Year Plan. `http://iepd.iipnetwork.org/policy/energy-development-plan-12th-five-year-plan`, accessed: 2014-10-10.

Inflation EU (2015). Historic Inflation China (CPI). `http://www.inflation.eu/inflation-rates/china/historic-inflation/cpi-inflation-china.aspx`, accessed: 2014-10-10.

IPCC (2007). Climate Change 2007: Working Group I: The Physical Science Basis. `http://www.ipcc.ch/publications\_and\_data/ar4/wg1/en/ch2s2-10-2.html`, accessed: 2014-05-04.

ISO (2006). ISO 14040: Environmental managementLife cycle assessment Principles and framework. Tech. rep., International Standards Organisation, Geneva.

Ison, R. and Russell, D. (2007). *Agricultural extension and rural development: breaking out of knowledge transfer traditions*. Cambridge University Press.

Jackson, T. (2011). *Prosperity without growth: Economics for a finite planet*. Routledge.

Jänicke, M. (2008). Ecological modernisation: new perspectives. *Journal of Cleaner Production*, **16**, 557–565.

Jeffery, S., Verheijen, F., van der Velde, M. and Bastos, A. (2011). A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agriculture, Ecosystems & Environment*, **144**, 175–187.

Jha, A. (2009). Biochar goes industrial with giant microwaves to lock carbon in charcoal. `http://www.theguardian.com/environment/2009/mar/13/charcoal-carbon`, accessed: 2012-04-27.

Jiang, D., Zhuang, D., Fu, J., Huang, Y. and Wen, K. (2012). Bioenergy potential from crop residues in China: Availability and distribution. *Renewable and Sustainable Energy Reviews*, **16**, 1377–1382.

Jin, S., Huang, J., Hu, R. and Rozelle, S. (2002). The creation and spread of technology and total factor productivity in China's agriculture. *American Journal of Agricultural Economics*, **84**, 916–930.

Jin, X., Xu, Q. and Huang, C. (2005). Current status and future tendency of lake eutrophication in China. *Science in China Series C: Life Sciences*, **48**, 948–954.

Jingjing, L., Xing, Z., DeLaquil, P. and Larson, E.D. (2001). Biomass energy in China and its potential. *Energy for Sustainable Development*, **5**, 66–80.

Johannesson, M. and Jönsson, B. (1991). Economic evaluation in health care: is there a role for cost-benefit analysis? *Health policy*, **17**, 1–23.

Jones, L.P., Tandon, P. and Vogelsang, I. (1990). *Selling public enterprises: a cost-benefit methodology*. MIT press.

Joseph, S. (2009). Socio-economic assessment and implementation of smallscale biochar projects. In J. Lehmann and S. Joseph, eds., *Biochar for Environmental Management*, 359–374, Earthscan, 1st edn.

Joseph, S. (2014). Personal communication.

Joseph, S., Graber, E., Chia, C., Munroe, P., Donne, S., Thomas, T., Nielsen, S., Marjo, C., Rutlidge, H., Pan, G., Li, L., Taylor, P., Rawal, A. and Hook, J. (2013). Shifting Paradigms on Biochar: Micro/Nano-structures and Soluble Components are Responsible for its Plant Growth Promoting Ability. *Carbon Management*, **4**, 323–343.

Ju, X.T., Xing, G.X., Chen, X.P., Zhang, S.L., Zhang, L.J., Liu, X.J., Cui, Z.L., Yin, B., Christie, P., Zhu, Z.L. and Zhang, F.S. (2009). Reducing environmental risk by improving N management in intensive Chinese agricultural systems. *Proceedings of the National Academy of Sciences of the United States of America*, **106**, 3041–6.

Kahneman, D. (2011). *Thinking, fast and slow*. Macmillan. Chicago.

Karlen, D.L. and Johnson, J.M.F. (2014). Crop Residue Considerations for Sustainable Bioenergy Feedstock Supplies. *BioEnergy Research*, **7**, 465–467.

Khan, S., Cao, Q., Zheng, Y.M., Huang, Y.Z. and Zhu, Y.G. (2008). Health risks of heavy metals in contaminated soils and food crops irrigated with wastewater in Beijing, China. *Environmental pollution (Barking, Essex : 1987)*, **152**, 686–92.

Kimetu, J.M., Lehmann, J., Ngoze, S.O., Mugendi, D.N., Kinyangi, J.M., Riha, S., Verchot, L., Recha, J.W. and Pell, A.N. (2008). Reversibility of Soil Productivity Decline with Organic Matter of Differing Quality Along a Degradation Gradient. *Ecosystems*, **11**, 726–739.

Kirkels, A.F. and Verbong, G.P. (2011). Biomass gasification: Still promising? A 30-year global overview. *Renewable and Sustainable Energy Reviews*, **15**, 471–481.

Knoblauch, C., Maarifat, A.A., Pfeiffer, E.M. and Haefele, S.M. (2011). Degradability of black carbon and its impact on trace gas fluxes and carbon turnover in paddy soils. *Soil Biology and Biochemistry*, **43**, 1768–1778.

Kung, C.C., McCarl, B.a. and Cao, X. (2013). Economics of pyrolysis-based energy production and biochar utilization: A case study in Taiwan. *Energy Policy*, **60**, 317–323.

Kuzyakov, Y., Subbotina, I., Chen, H., Bogomolova, I. and Xu, X. (2009). Black carbon decomposition and incorporation into soil microbial biomass estimated by 14C labeling. *Soil Biology and Biochemistry*, **41**, 210–219.

Laird, D.A. (2008). The charcoal vision: A win-win-win scenario for simultaneously producing bioenergy, permanently sequestering carbon, while, wh improving soil and water quality. *Agronomy Journal*, **100**, 178–181.

Lal, R. (2002). Soil carbon sequestration in China through agricultural intensification, and restoration of degraded and desertified ecosystems. *Land Degradation & Development*, **13**, 469–478.

Lal, R. (2004). Carbon emission from farm operations. *Environment international*, **30**, 981–90.

Langhelle, O. (2000). Why ecological modernization and sustainable development should not be conflated. *Journal of Environmental Policy & Planning*, **2**, 303–322.

Lashari, M.S., Liu, Y., Li, L., Pan, W., Fu, J., Pan, G., Zheng, J., Zheng, J., Zhang, X. and Yu, X. (2013). Effects of amendment of biochar-manure compost in conjunction with pyroligneous solution on soil quality and wheat yield of a salt-stressed cropland from Central China Great Plain. *Field Crops Research*, **144**, 113–118.

Lazarus, R.J. (2008). Super wicked problems and climate change: Restraining the present to liberate the future. *Cornell Law Review*, **94**, 1153.

Leach, M., Fairhead, J. and Fraser, J. (2012). Green grabs and biochar: Revaluing African soils and farming in the new carbon economy carbon economy. *The Journal of Peasant Studies*, **39**, 37–41.

Lee, D.J., Tipton, T. and Leung, P. (1995). Modelling cropping decisions in a rural developing country: a multiple-objective programming approach. *Agricultural Systems*, **49**, 101–111.

Lehmann, J. (2007). A handful of carbon. *Nature*, **447**, 143–144.

Lehmann, J. (2009). Biological carbon sequestration must and can be a win-win approach. *Climatic Change*, **97**, 459–463.

Lehmann, J. and Joseph, S. (2015). Biochar for environmental management: an introduction. In J. Lehmann and S. Joseph, eds., *Biochar for Environmental Management*, 1–14, Earthscan, 2nd edn.

Lehmann, J. and Steiner, C. (2009). *Amazonian Dark Earths: Wim Sombroek's Vision*. Springer Netherlands, Dordrecht.

Lehmann, J., Gaunt, J. and Rondon, M. (2005). Bio-char sequestration in soil: a new frontier. `http://soilcarboncenter.k-state.edu/conference/PowerPoint_files/Lehmann_Baltimore_05_files/frame.htm`.

Lehmann, J., Gaunt, J. and Rondon, M. (2006). Bio-char Sequestration in Terrestrial Ecosystems A Review. *Mitigation and Adaptation Strategies for Global Change*, **11**, 395–419.

Lenton, T.M. and Vaughan, N.E. (2009). The radiative forcing potential of different climate geoengineering options. *Atmospheric Chemistry and Physics Discussions*, **9**, 2559–2608.

Li, J. and Wang, X. (2012). Energy and climate policy in China's twelfth five-year plan: A paradigm shift. *Energy Policy*, **41**, 519–528.

Li, L.J., Wang, Y., Zhang, Q., Li, J.X., Yang, X.G. and Jin, J. (2008). Wheat straw burning and its associated impacts on Beijing air quality. *Science in China, Series D: Earth Sciences*, **51**, 403–414.

Li, R. and Leung, G.C. (2012). Coal consumption and economic growth in China. *Energy Policy*, **40**, 438–443.

Li, V. and Lang, G. (2010). China's Green GDP Experiment and the Struggle for Ecological Modernisation. *Journal of Contemporary Asia*, **40**, 44–62.

Li, W., Li, Q. and He, X. (2006). Progress in the study on returning crop stalks to the field (in Chinese). *Hunan Agricultural Sciences*, **1**, 46–48.

Li, X., Wang, S., Duan, L., Hao, J., Li, C., Chen, Y. and Yang, L. (2007). Particulate and trace gas emissions from open burning of wheat straw and corn stover in China. *Environmental science & technology*, **41**, 6052–8.

Liaoning Government (2014). Energy densities of standard coal. `http://www.ljg.gov.cn/bsfw/xzfw/200908/P020090820318383814214.doc`, accessed: 2014-09-23.

Lin, J.Y. (1992). Rural Reforms and Agricultural Growth in China. *The American Economic Review*, **82**, 34–51.

Lin, R. and Song, D. (2002). Utilizing status and problems of crop straw on Guangdong province (in Chinese). *Soil and Environmental Sciences*, **11**, 110.

Liu, C., Lu, M., Cui, J., Li, B. and Fang, C. (2014a). Effects of straw carbon input on carbon dynamics in agricultural soils: a meta-analysis. *Global Change Biology*, **20**, 1366–81.

Liu, G. and Huang, M. (2011). Fuel collecting radius and installed capacity of straw stalk power plant. *Electric Power Construction*, **32**, 72–75.

Liu, X., Zhang, Y., Han, W., Tang, A., Shen, J., Cui, Z., Vitousek, P., Erisman, J.W., Goulding, K., Christie, P., Fangmeier, A. and Zhang, F. (2013). Enhanced nitrogen deposition over China. *Nature*, **494**, 459–62.

Liu, X., Ye, Y., Liu, Y., Zhang, A., Zhang, X., Li, L., Pan, G., Kibue, G.W., Zheng, J. and Zheng, J. (2014b). Sustainable biochar effects for low carbon crop production: A 5-crop season field experiment on a low fertility soil from Central China. *Agricultural Systems*, **129**, 22–29.

Liu, Y., Wang, X., Xiong, Y., Tan, H. and Niu, Y. (2014c). Study of briquetted biomass co-firing mode in power plants. *Applied Thermal Engineering*, **63**, 266–271.

Lo, A. (2012). Carbon emissions trading in China. *Nature Climate Change*, **2**, 765–766.

Lombardi, L. (2003). Life cycle assessment comparison of technical solutions for CO2 emissions reduction in power generation. *Energy Conversion and Management*, **44**, 93–108.

Lu, W. and Zhang, T. (2010a). Life-cycle implications of using crop residues for various energy demands in China. *Environmental Science & Technology*, **44**, 4026–32.

Lu, W. and Zhang, T. (2010b). Supporting Information: Life-cycle implications of using crop residues for various energy demands in China. *Environmental Science and Technology*, 1–62.

Lugato, E., Vaccari, F.P., Genesio, L., Baronti, S., Pozzi, A., Rack, M., Woods, J., Simonetti, G., Montanarella, L. and Miglietta, F. (2013). An energy-biochar chain involving biomass gasification and rice cultivation in Northern Italy. *GCB Bioenergy*, **5**, 192–201.

Major, J., Rondon, M., Molina, D., Riha, S.J. and Lehmann, J. (2010). Maize yield and nutrition during 4 years after biochar application to a Colombian savanna oxisol. *Plant and Soil*, **333**, 117–128.

Mankins, J.C. (1995). Technology readiness levels. Tech. rep., NASA, White Paper, April.

Mann, M. and Spath, P. (2001). A life cycle assessment of biomass cofiring in a coal-fired power plant. *Clean Products and Processes*, **3**, 81–91.

Mann, M.K. and Spath, P.L. (1997). Life Cycle Assessment of a Biomass Gasification Combined Cycle System. Tech. Rep. December, National Renewable Energy Laboratory, National Renewable Energy Laboratory, Task No. BP811030, Golden, Colorado.

McHenry, M.P. (2009). Agricultural bio-char production, renewable energy generation and farm carbon sequestration in western australia: Certainty, uncertainty and risk. *Agriculture, Ecosystems & Environment*, **129**, 1–7.

McKendry, P. (2002a). Energy production from biomass (Part 1): Overview of biomass. *Bioresource technology*, **83**, 37–46.

McKendry, P. (2002b). Energy production from biomass (Part 2): Conversion technologies. *Bioresource technology*, **83**, 47–54.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L., Pulgar-Vidal, M., Welch-Devine, M., Peter Brosius, J., Coppolillo, P. and O'Connor, S. (2011). Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation*, **144**, 966–972.

Meadows, D., Meadows, D.L., Randers, J. and Behrens, W.W. (1972). The limits to growth. Tech. rep., New York, 102.

Millenium Ecosystem Assessment (2005). *Ecosystems and human well-being*, vol. 5. Island Press Washington, DC.

Minchener, A. (2008). Development of co-firing power generation market opportunities to enhance the EU biomass sector through international cooperation with China. Tech. rep., CH.EU.BIO, Project funded by the European Commission within the Sixth Framework Programme (2002-2006).

Ming, C., Lixin, Z. and Yishui, T. (2008). Analysis and evaluation on energy utilization of main crop straw resources in China (in Chinese). *Transactions of the Chinese Society of Agricultural Engineering*, **24**, 291–296.

Mol, A.P. (1996). Ecological modernisation and institutional reflexivity: environmental reform in the late modern age. *Environmental Politics*, **5**, 302–323.

Mol, A.P. and Spaargaren, G. (1993). Environment, modernity and the risk-society: the apocalyptic horizon of environmental reform. *International sociology*, **8**, 431–459.

Monbiot, G. (2009). Woodchips with everything. It's the Atkins plan of the low-carbon world. `http://www.guardian.co.uk/environment/2009/mar/24/george-monbiot-climate-change-biochar`, accessed: 2011-11-15.

Muldavin, J.S. (1997). Environmental degradation in heilongjiang: policy reform and agrarian dynamics in china's new hybrid economy. *Annals of the Association of American Geographers*, **87**, 579–613.

National Bureau of Statistics of China (2003). *China's Statistical Yearbook*. China Statistics Press, Beijing.

National Bureau of Statistics of China (2008). Area of Cultivated Land at Year-End by Region. `http://www.stats.gov.cn/tjsj/ndsj/2012/html/M1303e.htm`, accessed: 2012-04-17.

National Bureau of Statistics of China (2013). Statistical Data. `http://www.stats.gov.cn/english/statisticaldata/`, accessed: 2012-05-06.

NBSC (2012). China Statistical Yearbook 2012. Tech. rep., National Statistics Bureau of China, Beijing.

Neves, E., Petersen, J., Bartone, R. and Augusto da Silva, C. (2003). Historical and socio-cultural origins of Amazonian dark earth. In *Amazonian dark earths: origin, properties, management*, 29–50, Kluwer Academic Publishers.

Pan, G. (2008). Soil Organic Carbon Stock, Dynamics and Climate Change Mitigation of China (in Chinese). *Advances in Climate Change Research*, **5**, 1–7.

Pearce, D.W. (1993). *Blueprint 3: Measuring sustainable development*, vol. 3. Earthscan.

Phelps, J., Friess, D.a. and Webb, E.L. (2012). Win-win REDD+ approaches belie carbon-biodiversity trade-offs. *Biological Conservation*, **154**, 53–60.

Pratt, K. and Moran, D. (2010). Evaluating the cost-effectiveness of global biochar mitigation potential. *Biomass and Bioenergy*, **34**, 1149–1158.

Qu, C., Li, B., Wu, H. and Giesy, J.P. (2012). Controlling air pollution from straw burning in China calls for efficient recycling. *Environmental Science and Technology*, **46**, 7934–7936.

Quaak, P., Knoef, H. and Stassen, H. (1999). Energy from biomass: A review of combustion and gasification technologies. World Bank Technical Paper No. 422. Tech. rep., The World Bank, Washington D.C.

Rae, A. (2008). China's agriculture, smallholders and trade: Driven by the livestock revolution? *Australian Journal of Agricultural and Resource Economics*, **52**, 283–302.

Read, P. (2008). Biosphere carbon stock management: addressing the threat of abrupt climate change in the next few decades: an editorial essay. *Climatic Change*, **87**, 305–320.

Read, P. (2009). This gift of nature is the best way to save us from climate catastrophe. `http://www.theguardian.com/commentisfree/2009/mar/27/biochar`, accessed: 2012-03-21.

Rittel, H. and Webber, M.M. (1973). Planning problems are wicked. *Polity*, **4**, 155–169.

Roberts, K.G., Gloy, B., Joseph, S., Scott, N.R. and Lehmann, J. (2010). Life cycle assessment of biochar systems: estimating the energetic, economic, and climate change potential. *Environmental Science & Technology*, **44**, 827–33.

Robertson, G.P. (2000). Greenhouse Gases in Intensive Agriculture: Contributions of Individual Gases to the Radiative Forcing of the Atmosphere. *Science*, **289**, 1922–1925.

Rogers, E.M. (2010). *Diffusion of innovations*. Simon and Schuster.

Röling, N. and Kaimowitz, D. (1990). Making the link: Agricultural research and technology transfer in developing countries.

Rosen, D.H., Rozelle, S. and Huang, J. (2004). *Roots of Competitiveness: China's Evolving Agriculture Interests*. Peterson Institute for International Economics.

Saft, R.J. (2007). LCA Case Studies Life Cycle Assessment of a Pyrolysis / Gasification Plant for Hazardous Paint Waste. *Hazardous Paint Waste*, **12**, 230–238.

Sanchez, P.a. (2002). Ecology. Soil fertility and hunger in Africa. *Science (New York, N.Y.)*, **295**, 2019–2020.

Sang, T. and Zhu, W. (2011). China's bioenergy potential. *GCB Bioenergy*, **3**, 79–90.

Scarlat, N., Martinov, M. and Dallemand, J.F. (2010). Assessment of the availability of agricultural crop residues in the European Union: potential and limitations for bioenergy use. *Waste management (New York, N.Y.)*, **30**, 1889–97.

Scholz, S.M., Sembres, T., Roberts, K., Whitman, T., Wilson, K. and Lehmann, J. (2014). Biochar Systems for Smallholders in Developing Countries. Tech. rep., World Bank, Washington DC.

Scoones, I. (1998). *Sustainable rural livelihoods: a framework for analysis. Vol. 72.*. Brighton: Institute of Development Studies.

Scoones, I. (2009). Livelihoods perspectives and rural development. *Journal of Peasant Studies*, **36**, 171–196.

Seifritz, W. (1993). Should we store carbon in charcoal? *International Journal of Hydrogen Energy*, **18**, 405–407.

Selten, R. (1990). Bounded rationality. *Journal of Institutional and Theoretical Economics*, **146**, 649–658.

Shackley, S., Carter, S., Sims, K. and Sohi, S. (2011a). Expert Perceptions of the Role of Biochar as a Carbon Abatement Option with Ancillary Agronomic and Soil-Related Benefits. *Energy & Environment*, **22**, 167–188.

Shackley, S., Hammond, J., Gaunt, J. and Ibarrola, R. (2011b). The feasibility and costs of biochar deployment in the UK. *Carbon Management*, **2**, 335–356.

Shackley, S., Clare, A., Joseph, S., McCarl, B. and Schmidt, H.P. (2015). Economic evaluation of biochar systems: current evidence and challenges. In J. Lehmann and S. Joseph, eds., *Biochar for Environmental Management*, chap. 29, 813–852, Earthscan, 2nd edn.

Siegel, P.B. and Alwang, J. (2005). Poverty Reducing Potential of Smallholder Agriculture in Zambia: Opportunities and Constraints. *Africa Region, Working Paper Series No.85*.

Simon, G.L., Bumpus, A.G. and Mann, P. (2012). Win-win scenarios at the climatedevelopment interface: Challenges and opportunities for stove replacement programs through carbon finance. *Global Environmental Change*, **22**, 275–287.

Singh, N., Abiven, S., Torn, M.S. and Schmidt, M.W.I. (2012). Fire-derived organic carbon in soil turns over on a centennial scale. *Biogeosciences*, **9**, 2847–2857.

Smith, N.J.H. (1980). Anthrosols and Human Carrying Capacity in Amazonia. *Annals of the Association of American Geographers*, **70**, 553–566.

Smith, P., Gregory, P.J., Van Vuuren, D., Obersteiner, M., Havlík, P., Rounsevell, M., Woods, J., Stehfest, E. and Bellarby, J. (2010). Competition for land. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **365**, 2941–2957.

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E., Haberl, H., Harper, R., House, J., Jafari, M., Masera, O., Mbow, C., Ravindranath, N., Rice, C., Robledo Abad, C., Romanovskaya, A., Sperling, F. and Tubiello, F. (2014). 11. Agriculture, Forestry and Other Land Use (AFOLU). Tech. rep., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Sombroek, W.G. (1966). *Amazon soils*. Doctoral Dissertation, Centre for agricultural publications and documentation.

Somebroek, W. (1993). Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. *Ambio*, **22**, 417–426.

Song, R. and Lei, H. (2014). Emissions Trading in China: First Reports from the Field. `http://www.wri.org/blog/2014/01/emissions-trading-china-first-reports-field`, accessed: 2014-08-03.

Sparrevik, M., Field, J.L., Martinsen, V., Breedveld, G.D. and Cornelissen, G. (2013). Life cycle assessment to evaluate the environmental impact of biochar implementation in conservation agriculture in Zambia. *Environmental science & technology*, **47**, 1206–15.

Spokas, K.A. and Reicosky, D.C. (2009). Impacts of sixteen different biochars on soil greenhouse gas production. *Annals of Environmental Science*, **3**, 4.

Spokas, K.A., Cantrell, K.B., Novak, J.M., Archer, D.W., Ippolito, J.A., Collins, H.P., Boateng, A.A., Lima, I.M., Lamb, M.C., McAloon, A.J., Lentz, R.D. and Nichols, K.A. (2012). Biochar: a synthesis of its agronomic impact beyond carbon sequestration. *Journal of environmental quality*, **41**, 973–89.

Stern, N. (2006). The economics of climate change. Tech. rep., HM Treasury, London, UK.

Sun, Z.J. and Sun, Y.M. (2006). Situation and Development of Agricultural Residues as Energy Resource Utilization in Rural Areas in China. *Review of China Agricultural Science and Technology*, **8**, 6–13.

Tallis, H., Kareiva, P., Marvier, M. and Chang, A. (2008). An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 9457–9464.

Tang, H., Qiu, J., Van Ranst, E. and Li, C. (2006). Estimations of soil organic carbon storage in cropland of China based on DNDC model. *Geoderma*, **134**, 200–206.

Teixeira, P., Lopes, H., Gulyurtlu, I., Lapa, N. and Abelha, P. (2012). Evaluation of slagging and fouling tendency during biomass co-firing with coal in a fluidized bed. *Biomass and Bioenergy*, **39**, 192–203.

Thaler, R.H. and Sunstein, C.R. (2008). *Nudge*. Yale University Press. Chicago.

The Economist (2015). Ending apartheid. `http://www.economist.com/news/special-report/21600798-chinas-reforms-work-its-citizens-have-be-made-more-equal-ending-apartheid`, accessed: 2015-21-09.

The UK Government Office for Science (2011). Foresight: The Future of Food and Farming. Final Project Report. `https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/288329/11-546-future-of-food-and-farming-report.pdf`, accessed: 2014-04-10.

Thomas, A., Bond, A. and Hiscock, K. (2013). A GIS based assessment of bioenergy potential in England within existing energy systems. *Biomass and Bioenergy*, **55**, 107–121.

Thomsen, T., Hauggaard-Nielsen, H. and Bruun, E.W. (2011). The potential of Pyrolysis Technology in climate change mitigation Risø-R-Report. Tech. Rep. January, National Laboratory for Sustainable Energy, Roskilde Denmark.

Tilman, D., Socolow, R., Foley, J.A., Hill, J., Larson, E., Lynd, L., Pacala, S., Reilly, J., Searchinger, T., Somerville, C. *et al.* (2009). Beneficial biofuelsthe food, energy, and environment trilemma. *Science*, **325**, 270.

Tilman, D., Balzer, C., Hill, J. and Befort, B.L. (2011). Global food demand and the sustainable intensification of agriculture. *Proceedings of the National Academy of Sciences*, **108**, 20260–20264.

Toke, D. (2002). Ecological modernisation and gm food. *Environmental politics*, **11**, 145–163.

Trading Economics (2014). China Interest Rate. `http://www.tradingeconomics.com/china/interest-rate`, accessed: 2014-02-12.

Tumuluru, J.S., Sokhansanj, S., Wright, C.T., Boardman, R.D. and Yancey, N.A. (2011). A review on biomass classification and composition, co-firing issues and pretreatment methods. *ASABE Annual International Meeting, paper number 1110458*.

UK Ministry of Defence (2014). Technology Readiness Levels (TRLs) in the Project Lifecycle.

US Dept. of Energy (2000). Biomass cofiring: a renewable alternative for utilities. Tech. rep., National Renewable Energy Laboratory, Washington DC, USA.

Utterback, J. (1996). *Mastering the dynamics of innovation*. Harvard Business Press, Cambridge, MA, USA.

van Loo, S. and Koppejan, J. (2008). *The handbook of biomass combustion & co-firing*. Earthscan, London.

van Zwieten, L., Kimber, S., Downie, A., Morris, S., Petty, S., Rust, J. and Chan, K.Y. (2010). A glasshouse study on the interaction of low mineral ash biochar with nitrogen in a sandy soil. *Australian Journal of Soil Research*, **48**, 569–576.

Van Zwieten, L., Kimber, S., Morris, S., Downie, A., Berger, E., Rust, J. and Scheer, C. (2010). Influence of biochars on flux of nitrous oxide and carbon dioxide from ferrosol. *Soil Research*, **48**, 555–568.

Vanclay, F. (1992). The social context of farmers' adoption of environmentally sound farming practices. In G. Lawrence, F. Vanclay and B. Furze, eds., *Agriculture, Environment and Society*, Melbourne, Macmillan.

Vanclay, F. (2004). Social principles for agricultural extension to assist in the promotion of natural resource management. *Australian Journal of Experimental Agriculture*, **44**, 213–222.

Wang, J., Pan, X., Liu, Y., Zhang, X. and Xiong, Z. (2012a). Effects of biochar amendment in two soils on greenhouse gas emissions and crop production. *Plant and Soil*, **360**, 287–298.

Wang, M. (2008). Greenhouse gases, regulated emissions, and energy use in transportation (GREET) model, 1.8b. Tech. rep., UChicago Argonne, LLC: Chicago, IL.

Wang, Q., Geng, C., Lu, S., Chen, W. and Shao, M. (2013a). Emission factors of gaseous carbonaceous species from residential combustion of coal and crop residue briquettes. *Frontiers of Environmental Science & Engineering*, **7**, 66–76.

Wang, S.S., Zhou, D.Q., Zhou, P. and Wang, Q.W. (2011a). Carbon dioxide emissions, energy consumption and economic growth in China: A panel data analysis. *Energy Policy*, **39**, 4870–4875.

Wang, W., Liu, Y. and Zhang, L. (2013b). The spatial distribution of cereal bioenergy potential in China. *Global Change Biology Bioenergy*, **5**, 525–535.

Wang, X., Tan, H., Niu, Y., Pourkashanian, M., Ma, L., Chen, E., Liu, Y., Liu, Z. and Xu, T. (2011b). Experimental investigation on biomass co-firing in a 300MW pulverized coal-fired utility furnace in China. *Proceedings of the Combustion Institute*, **33**, 2725–2733.

Wang, X., Huang, J., Zhang, L. and Rozelle, S. (2012b). Mechanization service and agricultural productivity: Evidence from Smallholders in Jiangsu province. Tech. rep., Center for Chinese Agricultural Policy, Chinese Academy of Sciences, Beijing, Center for Chinese Agricultural Policy, Chinese Academy of Sciences, Beijing.

Wang, Y.J., Bi, Y.Y. and Gao, C.Y. (2010). The Assessment and Utilization of Straw Resources in China. *Agricultural Sciences in China*, **9**, 1807–1815.

Wang, Z.W., Lei, T.Z., Yan, X.Y., Li, Y.L., He, X.F. and Zhu, J.L. (2012c). Assessment and utilization of agricultural residue resources in henan province, china. *BioResources*, **7**, 3847–3861.

Weale, A. (1992). *The new politics of pollution*. Manchester University Press.

Weifang Naipute Gas Genset Co. Ltd. (2014). 35kW DEUTZ syngas Generator Quotation. `www.wfnpt.com.cn/www.wfnpt.com`, accessed: 2014-01-08.

Werkelin, J., Skrifvars, B.J., Zevenhoven, M., Holmbom, B. and Hupa, M. (2010). Chemical forms of ash-forming elements in woody biomass fuels. *Fuel*, **89**, 481–493.

West, T.O. and Marland, G. (2002). A synthesis of carbon sequestration , carbon emissions , and net carbon flux in agriculture : comparing tillage practices in the United States. *Agriculture, Ecosystems & Environment*, **91**, 217–232.

Whitman, T. and Lehmann, J. (2009). Biochar: One way forward for soil carbon in offset mechanisms in Africa? *Environmental Science & Policy*, **12**, 1024–1027.

Whitman, T., Nicholson, C.F., Torres, D. and Lehmann, J. (2011). Climate change impact of biochar cook stoves in western Kenyan farm households: system dynamics model analysis. *Environmental science & technology*, **45**, 3687–94.

Wicks, R. and Keay, M. (2005). Can coal contribute to sustainable development? *Energy & Environment*, **16**, 767–780.

Willock, J., Deary, I.J., Mcgregor, M.M., Sutherland, A., Edwards-jones, G., Morgan, O., Dent, B., Grieve, R., Gibson, G. and Austin, E. (1999). Farmers Attitudes, Objectives, Behaviors, and Personality Traits: The Edinburgh Study of Decision Making on Farms. *Journal of Vocational Behaviour*, **54**, 5–36.

Wing Chan, K. and Buckingham, W. (2008). Is China Abolishing the Hukou System? *The China Quarterly*, **195**, 582–606.

Woods, J., Williams, A., Hughes, J.K., Black, M. and Murphy, R. (2010). Energy and the food system. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **365**, 2991–3006.

Woods, W.I. and McCann, J.M. (1999). The Anthropogenic Origin and Persistence of Amazonian Dark Earths. *Yearbook. Conference of Latin Americanist Geographers*, **25**, 7–14.

Woolf, D., Amonette, J.E., Street-perrott, F.A. and Lehmann, J. (2010a). Sustainable biochar to mitigate global climate change: Supplementary information. *Nature communications*, **1**, 1–42.

Woolf, D., Amonette, J.E., Street-Perrott, F.A., Lehmann, J. and Joseph, S. (2010b). Sustainable biochar to mitigate global climate change. *Nature communications*, **1**, 56.

World Bank (2014). Pump price for diesel fuel (China). `http://data.worldbank.org/indicator/EP.PMP.DESL.CD`, accessed: 2014-03-04.

World Bank (2015). World Bank Database: Carbon dioxide emissions. `http://data.worldbank.org/indicator/EN.ATM.CO2E.KT/countries?page=6`, accessed: 2015-02-03.

World Commission on Environment and Development (1987). *Our Common Future*. Oxford University Press.

World Resources Institute (2014). Getting every ton of emissions right. `http://www.wri.org/sites/default/files/calculation\_spreadsheet\_of\_china\_regional\_grid\_emission\_factors.xlsx`, accessed: 2014-01-26.

Wu, L., Chen, J., Zhu, X., Xu, Y., Feng, B. and Yang, L. (2001). Straw-burning in rural areas of China: caused and controlling strategy (in Chinese). *China Population, Resources and Environment*, **11**, 110–112.

Xingang, Z., Zhongfu, T. and Pingkuo, L. (2013). Development goal of 30GW for Chinas biomass power generation: Will it be achieved? *Renewable and Sustainable Energy Reviews*, **25**, 310–317.

Xiong, J., Zhao, H., Zheng, C., Liu, Z., Zeng, L., Liu, H. and Qiu, J. (2009). An economic feasibility study of O 2 / CO 2 recycle combustion technology based on existing coal-fired power plants in China. *Fuel*, **88**, 1135–1142.

Yan, X., Ohara, T. and Akimoto, H. (2006). Bottom-up estimate of biomass burning in mainland China. *Atmospheric Environment*, **40**, 5262–5273.

Yang, S., He, H., Lu, S., Chen, D. and Zhu, J. (2008). Quantification of crop residue burning in the field and its influence on ambient air quality in Suqian, China. *Atmospheric Environment*, **42**, 1961–1969.

Yao, Z., Wang, S. and Jiang, X. (2001). The current situation and approach of return straw to field in suburb of Shanghai (in Chinese). *Agro-Environment and Development*, **3**, 40–41.

Yu, Z. (2003). The developing trend of resources treatment of crop stalk in Fuzhou city (in Chinese). *Fujian Environment*, **20**, 31–32.

Yuan, J., Xu, Y. and Hu, Z. (2012). Delivering power system transition in China. *Energy Policy*, **50**, 751–772.

Yuan, J., Xu, Y., Zhang, X., Hu, Z. and Xu, M. (2014). China's 2020 clean energy target: Consistency, pathways and policy implications. *Energy Policy*, **65**, 692–700.

Yuan, X. and Zuo, J. (2011). Transition to low carbon energy policies in Chinafrom the Five-Year Plan perspective. *Energy Policy*, **39**, 3855–3859.

Zhang, A., Cui, L., Pan, G., Li, L., Hussain, Q., Zhang, X., Zheng, J. and Crowley, D. (2010a). Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. *Agriculture, Ecosystems & Environment*, **139**, 469–475.

Zhang, A., Cui, L., Pan, G., Li, L., Hussain, Q., Zhang, X., Zheng, J. and Crowley, D. (2010b). Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. *Agriculture, Ecosystems & Environment*, **139**, 469–475.

Zhang, A., Bian, R., Pan, G., Cui, L., Hussain, Q., Li, L., Zheng, J., Zheng, J., Zhang, X., Han, X. and Yu, X. (2012a). Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: A field study of 2 consecutive rice growing cycles. *Field Crops Research*, **127**, 153–160.

Zhang, A., Liu, Y., Pan, G., Hussain, Q., Li, L., Zheng, J. and Zhang, X. (2012b). Effect of biochar amendment on maize yield and greenhouse gas emissions from a soil organic carbon poor calcareous loamy soil from central china plain. *Plant and Soil*, **351**, 263–275.

Zhang, F. and Zhu, Z. (1990). Harvest Indices of Chinese Crops. *Scientis Agricultura Sinica (in Chinese)*, **23**, 83–87.

Zhang, F., Cui, Z., Fan, M., Zhang, W., Chen, X. and Jiang, R. (2011). Integrated Soil:Crop System Management: Reducing Environmental Risk while Increasing Crop Productivity and Improving Nutrient Use Efficiency in China. *Journal of Environmental Quality*, **40**, 1051–1057.

Zhang, F., Chen, X. and Vitousek, P. (2013a). An experiment for the world. *Nature*, **497**, 33–35.

Zhang, J., Smith, K., Ma, Y., Ye, S., Jiang, F., Qi, W., Liu, P., Khalil, M., Rasmussen, R. and Thorneloe, S. (2000). Greenhouse gases and other airborne pollutants from household stoves in China: a database for emission factors. *Atmospheric Environment*, **34**, 4537–4549.

Zhang, L., Mol, A.P.J. and Sonnenfeld, D.A. (2007). The interpretation of ecological modernisation in China. *Environmental Politics*, **16**, 659–668.

Zhang, Q., Zhou, D., Zhou, P. and Ding, H. (2013b). Cost Analysis of straw-based power generation in Jiangsu Province, China. *Applied Energy*, **102**, 785–793.

Zhang, Q., Zhou, D. and Fang, X. (2014). Analysis on the policies of biomass power generation in China. *Renewable and Sustainable Energy Reviews*, **32**, 926–935.

Zhang, S., Gao, P., Tong, Y., Norse, D., Lu, Y. and Powlson, D. (2015). Overcoming nitrogen fertilizer over-use through technical and advisory approaches: A case study from Shaanxi Province, northwest China. *Agriculture, Ecosystems & Environment (DOI:10.1016/j.agee.2015.03.002).*

Zhang, W.F., Dou, Z.X., He, P., Ju, X.T., Powlson, D., Chadwick, D., Norse, D., Lu, Y.L., Zhang, Y., Wu, L., Chen, X.P., Cassman, K.G. and Zhang, F.S. (2013c). New technologies reduce greenhouse gas emissions from nitrogenous fertilizer in China. *Proceedings of the National Academy of Sciences of the United States of America*, **10**, 8375–8380.

Zhang, X., Wang, H., He, L., Lu, K., Sarmah, A., Li, J., Bolan, N.S., Pei, J. and Huang, H. (2013d). Using biochar for remediation of soils contaminated with heavy metals and organic pollutants. *Environmental Science and Pollution Research*, **20**, 8472–8483.

Zhang, X.P. and Cheng, X.M. (2009). Energy consumption, carbon emissions, and economic growth in China. *Ecological Economics*, **68**, 2706–2712.

Zhao, Z.Y. and Yan, H. (2012). Assessment of the biomass power generation industry in China. *Renewable Energy*, **37**, 53–60.

Zhao, Z.Y., Zuo, J., Fan, L.L. and Zillante, G. (2011). Impacts of renewable energy regulations on the structure of power generation in China A critical analysis. *Renewable Energy*, **36**, 24–30.

Zhu, Y. (2007). China's floating population and their settlement intention in the cities: Beyond the hukou reform. *Habitat International*, **31**, 65–76.

Zimmerman, A.R., Gao, B. and Ahn, M.Y. (2011). Positive and negative carbon mineralization priming effects among a variety of biochar-amended soils. *Soil Biology and Biochemistry*, **43**, 1169–1179.